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# Proceedings

# 11th Central Hardwood Forest Conference

## Columbia, Missouri
## March 23-26, 1997

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# 11th CENTRAL HARDWOOD FOREST CONFERENCE

Proceedings of a Meeting

Held at

The University of Missouri

Columbia, MO

March 23-26, 1997

Edited by

Stephen G. Pallardy, Robert A. Cecich, H. Gene Garrett, and Paul S. Johnson

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## FOREWORD

This conference is the eleventh in a series of biennial meetings that began in 1976 at Southern Illinois University. Other conferences have been hosted by Purdue University, University of Kentucky, University of Illinois, University of Tennessee, Southern Illinois University with the North Central Forest Experiment Station (NCFES), Pennsylvania State University with the Northeastern Forest Experiment Station (NEFES), Purdue University with NCFES and West Virginia University with the NEFES. The purpose of these conferences has been the same: to provide a forum for the exchange of information concerning the biology and management of Central Hardwood forests by forest scientists throughout the region. As with previous proceedings, a wide range of topics that represent the broad array of research programs in this region are presented. The information in this Proceedings is important to the long-term management of forest resources in this unique region.

## REVIEW PROCEDURES

Each manuscript was reviewed by two or three scientists with expertise closely aligned to the subject matter. After decisions concerning acceptance, reviews were returned to the senior author who revised the manuscript and resubmitted it in diskette form suitable for printing by the North Central Forest Experiment Station, USDA Forest Service. Papers were edited to a uniform format and type style; however, authors are responsible for the accuracy and content of their papers.

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# A RESOURCE AT THE CROSSROADS:

# A HISTORY OF THE CENTRAL HARDWOODS

Ray R. Hicks, Jr.[1]

Abstract: The Central Hardwood Forest is an oak-dominated deciduous forest that stretches from Massachusetts to Arkansas and occurs in hilly to mountainous terrain. It is the largest and most extensive temperate deciduous forest in the world.

During the past 20 million years or so, angiosperms have been gradually replacing gymnosperms as the dominant plant form on earth, and deciduous hardwoods are particularly adapted to the fluctuating seasonal climate and moderate rainfall associated with the Central Hardwood Region. As the glacial ice sheets retreated about 12-14 thousand years ago, forests re-invaded the region. Native peoples populated the area and practiced extensive burning to improve habitat for game and to aid in land clearing for agriculture. By the time the early Europeans arrived, about 450 years ago, the Native People had adopted a sedentary lifestyle and their population had increased to levels comparable to that of Western Europe. The first European explorers brought with them diseases that decimated the native populations, and for almost 200 years the central hardwood forest re-grew to become the "primeval forest" of legend.

By the late 18th century, settlement by subsistence farmers using European methods (draft animals and metal tools) was in full swing. Much of the tillable land in the central hardwood region was cleared, including substantial areas of steep and hilly land. The industrial revolution in the post-Civil War period ushered in a trend away from subsistence farming and marginal land was abandoned to revert back to forest.

The last 100 years has seen an era of exploitive logging in the central hardwood region, followed by a re-growing forest. Human influences have predominated in shaping the present forest and these include logging, burning, grazing, fire control, wildlife management and pest introductions. In the past 30 years, societal attitudes toward land and forests has had a profound effect on governmental policy as it relates to forestry. The maturing central hardwood forest, mostly owned by private individuals, is rapidly becoming a **resource at the crossroads**.

## INTRODUCTION

The historical developments that shaped the central hardwood region are significant in understanding how the present forests came to be and, as always, knowledge of history enables us to learn from past successes and failures. Thus, history provides not only the background, but also the context for future management. For purposes of this discussion, the history of the central hardwoods will focus on the period after the last ice age (the period of human impact). Because the particular forest that exists today is mostly a result of events that have occurred during the life of the trees that presently comprise it, this period will be emphasized.

The overriding factors that predispose a given geographic region to a particular vegetation type are the climate and soil (site), and this has been true throughout the hundreds of millions of years that vegetation has existed on earth. Furthermore the prevailing vegetation creates habitat for animal forms which are adapted to it, and they, in turn, have a biofeedback relationship with the vegetation. This type of association is the basis for what is called an **ecosystem** (Odum 1971).

[1] Professor of Forestry, West Virginia University, Morgantown, WV 26506-6125

## EVOLUTION AND THE DECIDUOUS FOREST

The ability of plants to capture solar energy by using readily available elements (carbon, hydrogen, oxygen, etc.) has been the key to all biological development on earth. Fossil evidence is our window to the past, and although incomplete, it appears that this process of **photosynthesis** first developed in primitive marine algae between 400 and 500 million years ago (Eyre 1963). From these evolved the primitive land plants that occupied the earth's surface prior to the Carboniferous Period (more than 200 million years ago). During the Carboniferous Period (215-280 million years ago), ferns and their relatives dominated the earth. Around 190 million years ago, the gymnosperms began to take over as the principal vegetation, and they have given way to the dominance of angiosperms (including deciduous hardwoods) during the last 50 or so million years. The angiosperms, though relatively recent in their development, have shown a remarkable ability to adapt to the changing conditions of the earth. From this point of view, it might be said that the central hardwoods represent the cutting edge of evolutionary change, as compared to the boreal conifer forest.

Perhaps the most recent (and most significant) episode of climatic and vegetational change to affect the distribution of present-day vegetation was the so-called **ice age**. The continental glaciers extended down through North America into southern Illinois, and a zone of tundra extended well down into much of the present central hardwood region (Davis 1983). As the Wisconsin ice sheet began to retreat northward about 14,000 years ago, a boreal forest type replaced the tundra and vestiges of this forest still exist as the high-elevation spruce and spruce-fir communities in the Appalachians (Maxwell and Davis 1972). The deciduous forest community has gradually become the dominant vegetation of the region and now forms the most extensive forest of this type in the world.

## IMPACT OF THE NATIVE PEOPLE ON THE PRE-COLUMBIAN FOREST

As important as the ice age, and its retreat, has been, the impact of humans has probably been just as significant in shaping the central hardwood forest. The indigenous people occupied the central hardwood region for at least 10,000 years before Europeans came (Lesser 1993). In the southern portion of the region, human occupancy probably dates back as long as 14,000 years ago owing to the more hospitable climate farther south (Buxton and Crutchfield 1985). Archaeologists classify the human occupancy of North America into Late Prehistoric, Mississippian, Woodland, Archaic and Paleo periods (Lesser 1993). Table 1 shows the chronology of these periods and outlines their cultural lifestyle as well as broad climatic and vegetational environments (Buckner 1992). Prior to the Woodland period, indigenous peoples were nomadic hunter-gatherers. Their influence on the forest during this period was most likely through the setting of fires. Accidental human-caused fires no doubt occurred during this time, and perhaps intentional fires were set to drive out game or to retain habitat. MacCleery (1992) speculates that the early Europeans' observations of abundant game (bison, elk, etc.) in the central hardwood region indicates a greater proportion of grassland occurred than currently exists, which implies that burning probably took place. Such burning, most likely would have been the work of the indigenous population.

Around 1,000-500 B.C., the Native Americans adopted a more sedentary lifestyle, and at some point during that time, they began the practice of agriculture (Swanton 1979). They also constructed elaborate burial mounds which can be found in places, such as Moundsville, West Virginia; Marietta and Hillsboro, Ohio; East St. Louis, Illinois; and Cartersville, Georgia. During this agricultural phase, which extended into the Late Prehistoric Period until European contact (about A.D. 1600), the Native Americans generally lived in permanent villages, often on the level land of river floodplains (Davis 1978). The village would often have a structural layout with houses of thatch or pole construction surrounded by gardens and fields. During this time, the Native Americans used tools as well as fire to clear land. MacCleery (1992) states the case as follows: " The south was dominated by fire-created forests, such as long-leaf pine savannas on the Coastal Plain and Piedmont. The hardwood forests of the Appalachian Mountains were also burned frequently by native peoples. Virginia's Shenandoah Valley—the area between the Blue Ridge Mountains and the Alleghenies—was one vast grass prairie. Native peoples burned the area annually." MacCleery further states: "On the western fringe of the eastern forest, fire-dominated forests, such as oak and oak-pine savannas, covered tens of millions of acres." Martin and Houf (1993) indicated that remnants of so-called "balds" still exist in the Ozarks of southwestern Missouri. These grassy glades usually occur on dolomitic limestone soils, and require periodic fires for their maintenance. They tolerate intermittent grazing, as would have occurred

when migratory herds of bison occupied the area, but under year-round grazing imposed by the European settlers, eastern redcedar began to overtake the glades. Beilman and Brenner (1951) use several lines of evidence to indicate that extensive forest cover is a relatively recent development in the Ozark region. They refer to Houck's (1908) account of Ferdinand DeSoto's observations of the region in 1541 where he described fertile alluvial bottoms planted in maize, pecans, plums and mulberry trees. The open, park-like countryside was in contrast to the hardwood and pine forests that exist today. Another description of park-like vegetation in the Ozark region was provided by the explorer Coronado, where he noted the hunting of buffalo by the Osage Indians. Although substantial prairie-like areas probably occurred throughout the central hardwood region, Steyermark (1959) indicates that even within the Ozark region, where grasslands were common, substantial areas of hardwood forests existed prior to European settlement.

Table 1. Chronology of Native People from Buckner (1992)

| Cultural Event | Date | Years ago | Climatic/vegetational stages |
|---|---|---|---|
| HISTORIC PERIOD: | | | |
| Modern Times. Settlement Times. | 2000 | 0 | Man-made forests widespread. |
| High Indian mortality. | | | Exploitation of forests and soil. |
| America discovered. | 1500 | 500 | Indian impacts (cultivation and fire) |
| MISSISSIPPIAN PERIOD: | | | mold forest character. |
| Indian cultures largely agrarian, | | | |
| large palisades. | 1000 | 1,000 | |
| WOODLAND PERIOD: | | | |
| Pottery | 500 | 1,500 | |
| Corn cultivated; bow and arrow | | | |
| | AD | | Northern pines had moved into Canada |
| Burial mounds | 0 | 2,000 | while southern pines had moved into |
| | BC | | Tennessee - their present distributions. |
| ARCHAIC PERIOD: | 1000 | 3,000 | |
| Marked increase in Indian | | | |
| population; exchange with | | | |
| other regions. | 2000 | 4,000 | Sea level rises to modern position. |
| Beginnings of cultivation with | | | |
| fire as the only feasible tool | | | |
| for land clearing. | 3000 | 5,000 | "Southern pine rise" results in marked |
| | | | increase in dominance of southern |
| | | | pines in SE. |
| Archeological evidence that | 4000 | 6,000 | |
| Archaic Indians used total | | | |
| landscape of So. Appalachians. | | | |
| | 5000 | 7,000 | Increased summer warmth and drought. |
| | 6000 | 8,000 | |
| | 7000 | 9,000 | Central hardwood oak-hickory forests |
| | | | became established. |
| | 8000 | 10,000 | Periglacial climate extended as far south |
| PALEO-INDIAN PERIOD: | | | as an east-west line thru Ashville, NC |
| Largely hunting/gathering tribes; | | | |
| fire was an available tool. | 9000 | 11,000 | Temperate, deciduous forests replace Jack |
| | | | pine/spruce/fir/larch in TN and NC. |
| First evidence of humans in SE | 10000 | 12,000 | |
| | | 15,000 | Jack pine, spruce and fir are the primary |
| | | | forest types as far south as Tennessee. |
| | | 18,000 | FULL GLACIAL MAXIMA |

Because the indigenous people tended to settle on floodplains of major rivers, their influence was probably greatest in such areas. Therefore, in many of the regions that are dominated by uplands where rivers have narrow floodplains, such as interior West Virginia, western Kentucky, southeastern Virginia and into the Cumberland Plateau of Tennessee, the evidence of Native American influence is more limited. However, their use of fire may have enabled them to exert an influence far removed from their settlements. Cronon (1983), with reference to the American Indian influence on the landscape states: "It is tempting to believe that when the Europeans arrived in the New World they confronted Virgin Land, the Forest Primeval, a wilderness which had existed for eons uninfluenced by human hands. Nothing could be farther from the truth—Indians had lived on the continent for thousands of years, and had to a significant extent modified its environment to their purposes." The population density of Native Americans is still being debated, with estimates ranging from 1 million to more recent higher estimates, up to 18 million (Buckner 1992). At higher levels, their impact on the forest would have been greater, but in any event, they most certainly had a significant effect on the central hardwood region.

## POST EUROPEAN SETTLEMENT TO 1860

The first Europeans in North America, apart from settlers in colonial villages along the Atlantic seaboard, were essentially hunter/trapper/explorers. They had a limited direct impact on the forest, but their indirect impact was profound. They brought very little with them in the way of equipment, livestock or supplies, but they did bring something that proved to be more significant than any of these—disease. As Williams (1989) points out, the epidemic of disease introduced by the Europeans affected the native peoples to such a degree that between 1520 and 1700, there was a return to a more forested landscape in North America (Buckner 1992). It was this forest that most historians refer to as the "impenetetrable anchient forests." Early explorers kept few records of their observations, but they opened the way for others including botanists, surveyors, etc. Britain offered huge grants of land to individuals, such as Lord Fairfax and William Penn, to encourage settlement of the new world and to gain control of the land from the native population. By 1600 A.D. waves of European immigrants, looking for a better life and an opportunity to escape the feudal system, made their way to the new world.

Most early settlers were subsistence farmers, a technology the Europeans had adopted 1,000 years earlier (Blethen and Wood 1985). The European model had proved very successful in their homeland, therefore, the immigrants set about clearing the North American forest to plant their crops and graze their livestock, in much the same way as they had "tamed" the European landscape about 1,000 years earlier. They brought with them two things the native people lacked—metal tools (ploughs, axes) and draft animals. Their attitude was that the forest represented an obstacle to be conquered. The forest also harbored wolves, mountain lions and bears which posed a threat to domesticated livestock, such as sheep and hogs.

Although the forest was a challenge to be overcome, forests also provided many of the early settlers' needs. For example, nearly every family lived in a wooden house and kept their livestock in a wooden barn (usually log construction). Additionally, wood was the primary fuel for heating. It was not uncommon for a family to use 20-40 cords of wood annually to heat and cook (MacCleery 1992). Wood was used for fencing. MacCleery (1992) estimated that it required about 8,000 fence rails to enclose a 40-acre square field, and by 1850 there were about 3.2 million miles of rail fence in the United States (mostly in the East). Potash was another important product from American forests in the early- to mid-eighteenth century. The demand for potash in Britain was great during that time. It was used as a soil amendment and as an ingredient in industrial processing (Williams 1989). Hardwoods, especially oaks and maples, produced the highest amounts of potash per unit wood burned and were preferred by potash producers.

From the middle 1700's into the 1800's, many naturalists trekked across North America in pursuit of unknown, unnamed or undescribed species of plants and animals. Some were students of Carolus Linnaeus, the Swedish botanist. Others, such as Andre Michaux (1805) and John Bartram (1751) and his son William (1791) published detailed accounts of their travels through the central hardwood region. From their descriptions and others, such as surveyors' field notes, we can get an idea of what the forest was like at that time.

By this time the Native American populations had been decimated by disease, and the deciduous forests had been virtually free from their effect for between 150 to 250 years. These observers wrote about vast forests of hardwoods. Stephenson (1993) quotes Diss Debar's description of the high-elevation forests of West Virginia on the "table-lands" of the Cheat and Greenbriar mountains: "Here, also neither Oak, Poplar nor Hickory are to be found, but in their room thrive noble specimens of Sugar Maple, Ash, Beech, Birch, Wild Cherry and Black Walnut." But observers who wrote about the Ridge and Valley and Appalachian Plateau forests of lower elevation noticed an abundance of oaks, hickories, maples, and yellow-poplar. Using the frequency of mention as an index to species abundance, from John Bartram's (1751) travels through Pensilvania [sic] to Onandaga, the following ranking of species in order of importance was developed (Table 2). Although the age and size of trees and extent of the forest was greater in 1750 than it is today, the species mix looks fairly typical of present-day stands in the northern part of the central hardwood region.

Table 2. Summary of frequency of mention by species (or species groups) from Bartram's (1751) trip through Pennsylvania

| Species | Number times mentioned | Ranking |
|---|---|---|
| White oak and black oak | 25 | 1 |
| White pine | 12 | 2 |
| Chestnut | 10 | 3 |
| Spruce | 10 | 3 |
| Hickory | 8 | 4 |
| Sugar maple | 8 | 4 |
| Linden | 7 | 5 |
| Pitch pine | 7 | 5 |
| Elm | 6 | 6 |
| Beech | 6 | 6 |
| White walnut (butternut?) | 6 | 6 |
| Birch (yellow?) | 5 | 7 |
| Poplar (yellow-poplar?) | 4 | 8 |
| Ash | 4 | 8 |
| Sugar birch (black?) | 3 | 9 |
| Great magnolia (cucumber?) | 3 | 9 |
| Locust (black?) | 2 | 10 |
| Walnut (black?) | 1 | 11 |
| Hophornbeam | 1 | 11 |
| Plane (sycamore?) | 1 | 11 |

The early immigrants settled along the Atlantic Coast and generally used rivers as a primary means of transportation. Westward progress of settlement was relatively slow from 1600 well into the 1700's, mostly due to the lack of a transportation system. The Fall Line of the Peidmont marked the limit of navigability on most eastern rivers; thus, many settlements were established at this position along the frontier. Innovative feats of engineering created water transportation systems, such as the C and O Canal, brainchild of George Washington, but it was only possible to proceed from the Chesapeake Bay to Cumberland, Maryland. The Allegheny Front proved too formidable for the technology of the day. It was the railroads that finally opened the land beyond the Alleghenies. Even though roads (such as the Drover's Road in North Carolina) were used during this time, they were not suitable for transporting heavy materials. Lewis (1995) indicated that it was the construction of the railroads into West Virginia's back counties that allowed a "timber boom" to occur and transformed subsistence farming into "commercial" agriculture in the state.

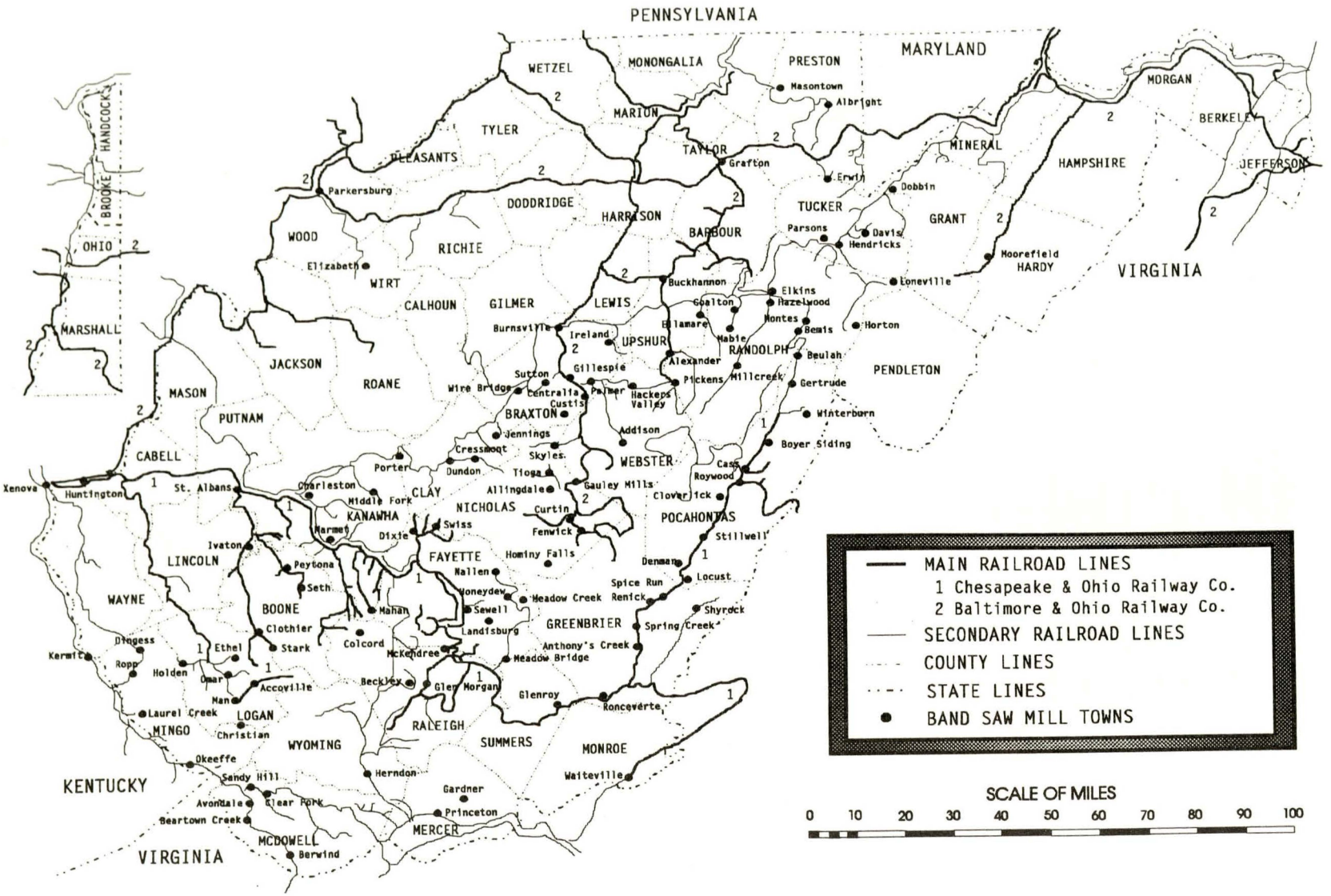



Figure 1. Clarkson's (1964) map of West Virginia railroads in 1917 and band saw mill towns, 1875-1920.

A fledgling iron industry was developing in the eastern United States, especially in the Appalachian Plateaus. Low-grade iron ore was processed in stone furnaces using local limestone for purification and charcoal for heating. Charcoal was produced by clearing patches of forest 1-3 acres in size (Clatterbuck 1990). Luther (1977) estimated that the 11 furnaces operating in the mid-1800's on the Highland Rim of Middle Tennessee required 375 square miles (240,000 acres) of timber to support them. Similar operations in West Virginia, Pennsylvania and elsewhere combined would have consumed several million acres of forestland in the central hardwood region.

## THE INDUSTRIALIZATION PERIOD, 1861-1929

As is often the case in a post-war era, the Reconstruction period following the Civil War ushered in a period of sweeping change. The steam engine, which could be moved from place to place, was taking the place of water power for milling. The factories of the North were being converted to peacetime production. And the development of a massive rail transportation system was well under way, spurred on initially to supply troops in the war (Figure 1). With the end of slavery, many large plantation farms were unprofitable. Many subsistence farmers throughout the central hardwood region gave up farming for a more lucrative lifestyle to work in factories, mines or logging camps. Extensive tracts of land were purchased and consolidated by large timber or mineral companies as a speculative enterprise (Eller 1985). The migration of farmers to logging, milling or mining camps was taking place nation wide (Figure 2), but was more pronounced in areas, such as the central hardwood region, where small subsistence farming predominated.

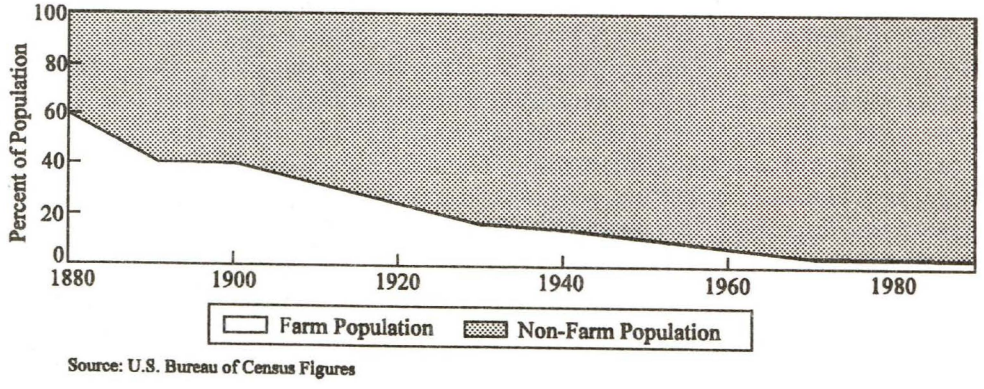
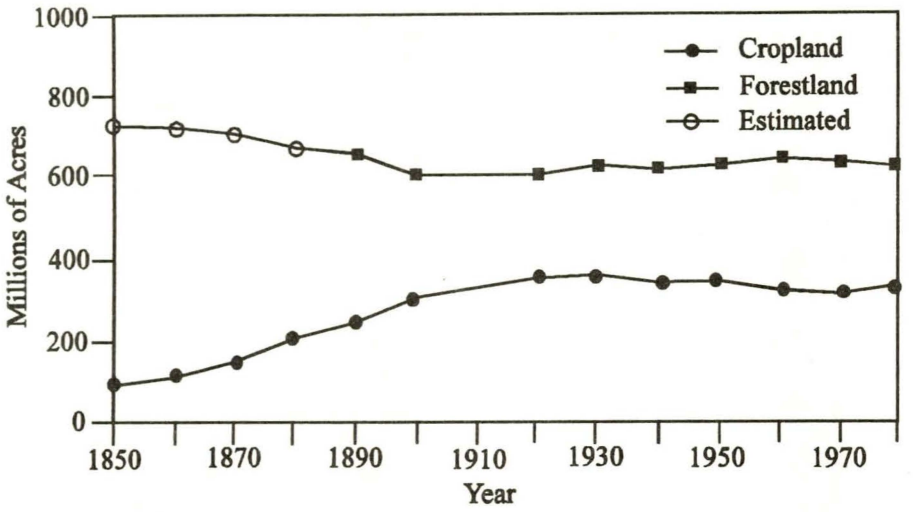



Figure 2. Farm and non-farm population, 1880-1988, from MacCleery (1992).

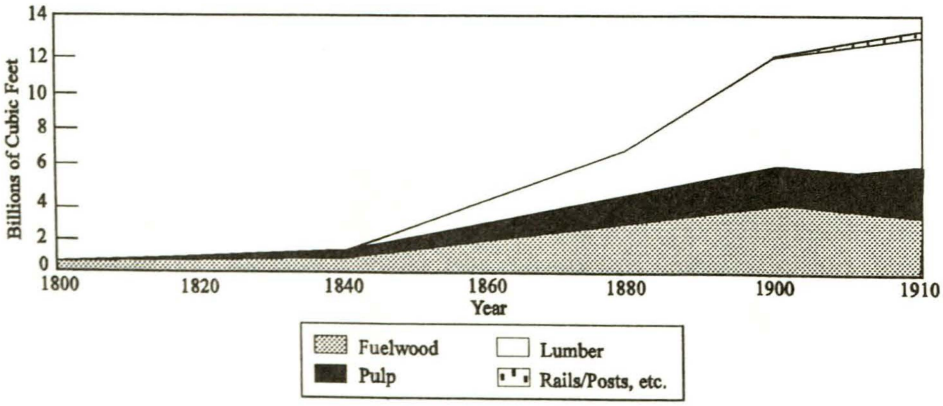



Source: RPA Technical Report, RM-175, USDA-Forest Service, 1989.

Figure 3. Cropland vs forest area, 1850-1980, from MacCleery (1992).

In addition to the emergence of America as an industrial nation, the Industrial Revolution brought about significant changes in agriculture. Tractors powered by steam, and later gasoline, replaced draft stock, and the use of fertilizers and genetically improved plants and animals increased production per unit area. Up until about 1908, as the population of the United States grew, more land needed to be placed under cultivation, and, conversely, land was taken out of forest production (Figure 3). The amount of forestland and cropland has stabilized with the small additional withdrawals of forestland coming from urbanization and rights-of-way for highways, pipelines and power transmission. Mechanization of agriculture also caused a shift in the type of land that could be easily cultivated. Much of the steeper land within the central hardwood region became submarginal and was abandoned. The capital investment required to buy large machinery put agriculture on more of a business level than that of a homesteading operation.

Several secondary effects of the industrialization period on the central hardwoods were also felt. For example, the factories, machines and vehicles needed fuel which promoted rapid development of the fossil fuel industry. Coal, oil and gas production was spurred on throughout the central hardwood region. Forests were withdrawn from production by surface mining, roads, well sites, pipelines, etc. In addition, the booming economy required wood for construction of factories, new housing for factory workers (either immigrants or relocated farmers), mine timbers, etc.




Source: Frederick &Sedjo, RFF (1991)

Figure 4. Domestic production, 1800-1910.

The booming economy ushered in by the industrial revolution and the vast resource base in North America swept over the timber industry as well. According to Frederick and Sedjo (1991), production of forest products tripled in the United States between 1860 and 1910 (Figure 4), mostly as a result of increased production of lumber. Lumber production in West Virginia, where hardwoods predominated, showed a similar trend (Figure 5). The economic boom in North America for consumer goods continued until the beginning of the Great Depression, but the timber boom had peaked in the central hardwood region by 1920, mostly as a result of overcutting. Ahern (1928) in his booklet "Deforested America" stated the situation as follows: "In 1919 the annual drain on our forest resources was estimated at four times the annual growth." Other estimates, although less dramatic, show the same trend where drain exceeded growth during this period (Figure 6).

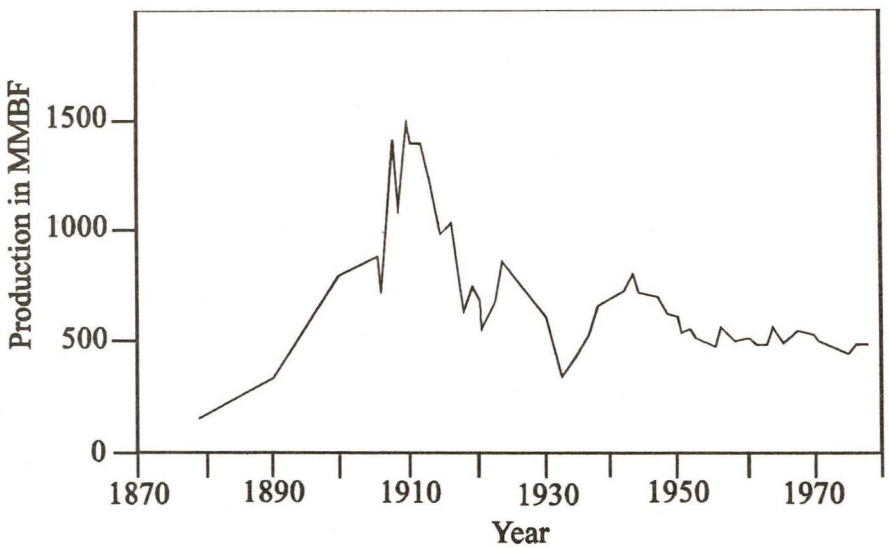



Figure 5. West Virginia lumber production from Zinn and Jones (1984).

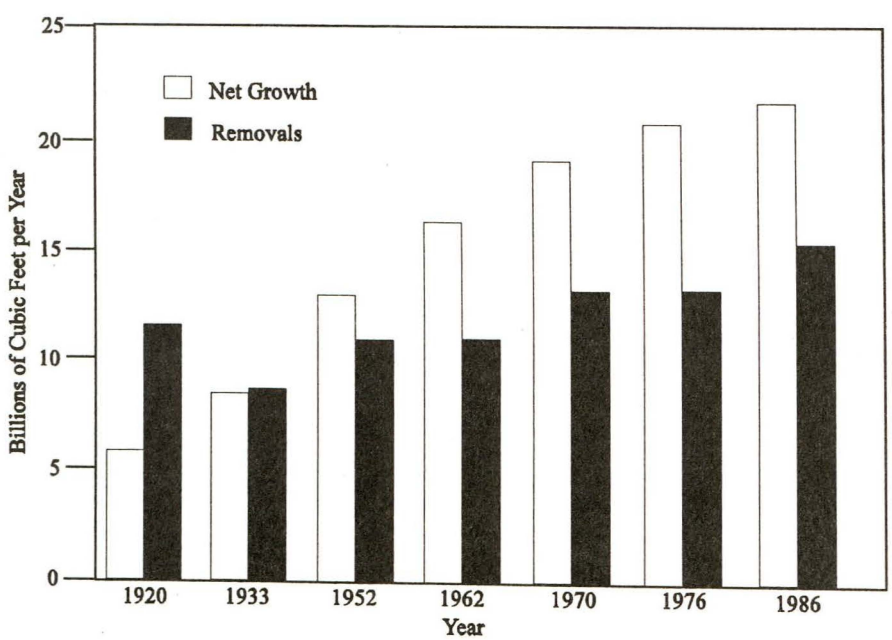



Figure 6. Timber growth/removals from MacCleery (1992).

The history of the North American logging boom is the subject of several books (Blackhurst 1954, Brown 1923, Clarkson 1964, Fries 1951), and like similar events, such as the California Gold Rush, the timber boom is surrounded by a certain amount of folklore as well as fact. But, in any event, it had a more dramatic effect on our present day central hardwood forest than any single event. Almost all the forests that had any merchantable value were cut over during that period. Only small pockets remained, such as found in the Joyce Kilmer Memorial Forest in North Carolina (Lorimer 1976). The use of steam engines for logging and steam donkeys for skidding exacerbated the effect in causing numerous fires which burned repeatedly through the logging slash and forest floor, exposing the mineral soil to erosion. Eller (1985) quotes Thomas Wolfe of Ashville, North Carolina, as follows: "The great mountain slopes and forest had been ruinously detimbered; the farm-soil on the hillsides had eroded and washed down; high up, upon the hills one saw raw scars of old mica pits, the dump heaps of deserted mines. . . . It was evident that a huge compulsive greed had been at work." Boom towns, such as Davis and Spruce, West Virginia; Tellico Plains and Gatlinburg, Tennessee; and Fontana, North Carolina, developed rapidly; some are completely gone today, and others turned to other industry or tourism to survive. But in spite of the apparent devastation, the forests of the central hardwood region started to regrow, resulting from natural regeneration (stored seed, seed distributed by wind or animals, seedling sprouts or stump sprouts). This period of recovery is the focus of the following section.

## THE REGROWING FOREST, 1930-1996

Well before 1930, several persons in positions of leadership in American forestry had expressed grave concern about the state of the forest in North America. Gifford Pinchot, generally regarded as the father of American forestry, stated his opinion in Ahern's (1928) publication, as follows: "Forest fires are steadily growing worse in America, and fire prevention is absolutely indispensable. But the axe, carelessly used is the mother of forest fires. The axe and not fire is our greatest danger. Until the axe is controlled there can be no solution of the fire problem, or of the problem of forest devastation in America." Pinchot, who was a consummate politician, had been trained in forestry in Europe. His first job had been as a forester on the huge Vanderbilt estate near Asheville, North Carolina, in 1892 (later to become the site of the Biltmore Forest School), but his greatest accomplishments were in the political arena. He became a Washington political insider and the first Chief of the USDA, Forest Service. He was instrumental in establishment of the Yale School of Forestry and the Society of American Foresters. Pinchot's influence on the central hardwood forest came mostly through the establishment of the eastern National Forests. Earlier, Pinchot's influence on then President Theodore Roosevelt had resulted in substantial additions to the national forest reserve between 1905-1909. About 80 million acres of federal land in the western states had been set aside as forest reserves as Pinchot and others became increasingly concerned about overcutting in the East.

But the eastern lands had been purchased by private owners and were not so easily annexed by the federal government. The breakthrough came in 1911 with passage of the "Weeks Act" which authorized the federal government to purchase lands in eastern America. Initially, the purpose for acquiring the land was to protect headwaters of navigable streams. Lands which formed the nucleus of many of our current National Forests were purchased under the Weeks Act authority. Almost all the initial purchases were cut-over, and often burned-over, lands belonging to large corporate owners and were bought for prices of $2 to $4 per acre; hence, they were called "the lands that no one wanted."

In 1924, the Clarke-McNary Act added the production of timber to the mandate for National Forest lands. By 1930, the Great Depression was in full swing, and land prices, especially the cut-over forest lands, were extremely low, as corporate owners often viewed the land as a tax burden with no foreseeable harvest of timber in sight. For example, in 1933 a purchase of almost 330,000 acres for the Monongahela National Forest was made at an average cost of $3.43 per acre (USDA Forest Service 1970). From these beginnings, the eastern National Forests developed (Figure 7), many within the central hardwood region.

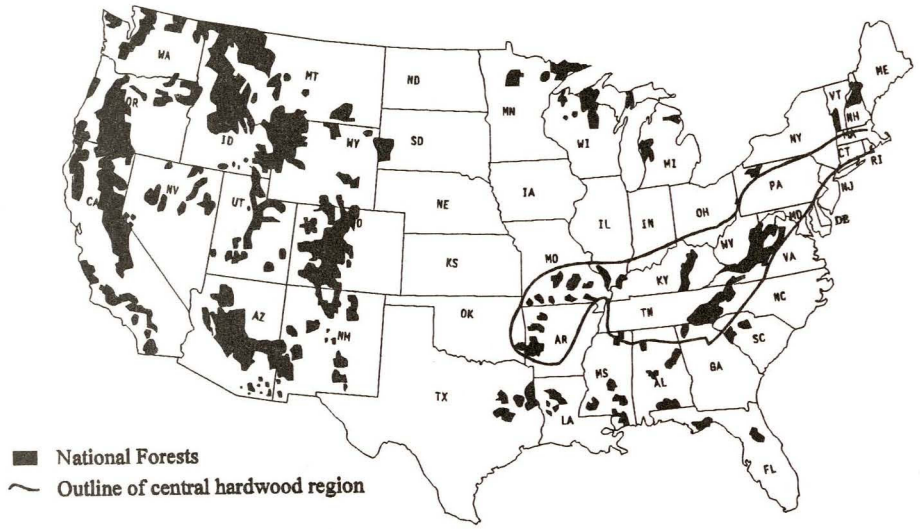



Figure 7. Distribution of National Forests.

The Weeks and Clark-McNary Acts, in addition to mandating the acquisition of land, also provided funding to states for fire control. The Forest Service also became the lead agency in promoting forest fire control at the federal level. The results came slowly at first, but by the 1960's, forest fires which consumed an average of 40-50 million acres in the 1930's were reduced to 2-5 million acres (USDA Forest Service 1987) (Figure 8). Although the central hardwood region is not as fire prone as conifer-dominated regions of the country, fire has always been a significant disturbance in the oak forests of the central hardwood region (Van Lear and Waldrop 1989). For example, in 1987 more than 50% of the land area in Mingo County, West Virginia burned over (Hicks and Mudrick 1993) (Figure 9).

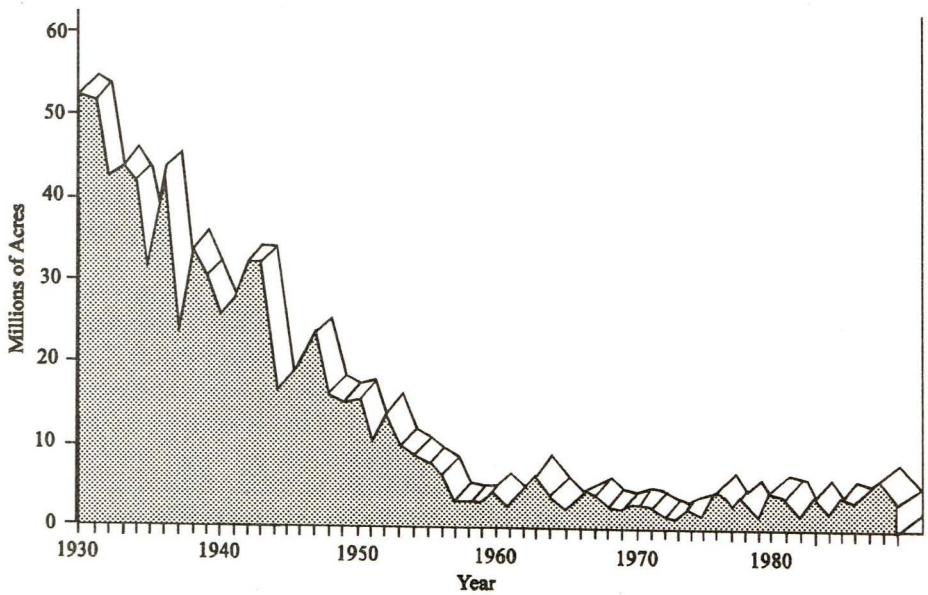



Figure 8. US wildfire trends, from MacCleery (1992).

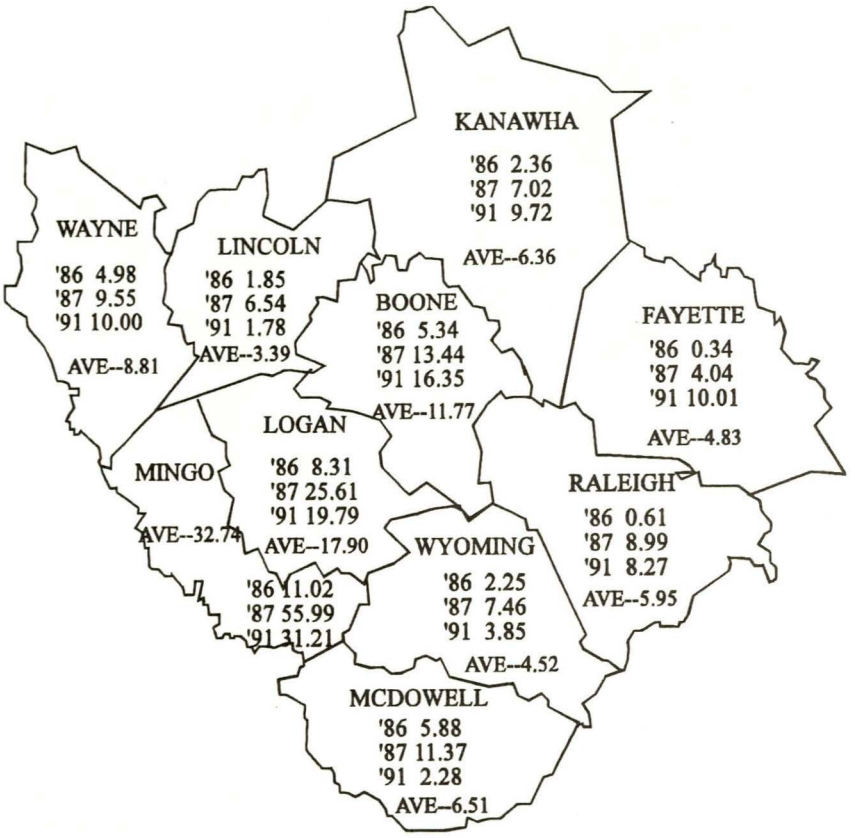



Figure 9. Percentage of West Virginia counties burned in 1986, 1987 and 1991.

In 1933, President Franklin D. Roosevelt signed the Civilian Conservation Corps bill into law. The CCC was part of Roosevelt's "New Deal," intended to solve two problems (unemployment and environmental destruction). It was estimated that nearly 250,000 young men who were of employment age, were out of work during the 1930's and Major Stuart, Chief of the USDA Forest Service, suggested that the army be used to build camps and generally administer the program. But unemployed civilians made up the work force. Many CCC camps were established within National Forests and their activities included planting of trees, fire control, watershed improvements, road improvement and construction of recreational facilities. Although their impact on today's central hardwood forests are largely undetectable, they planted millions of trees (mostly conifers) on denuded lands, and many of the recreational structures they built are still usable today, almost 60 years later.

The Depression drove more people from marginal farms, and the trend in reversion of farmland to forest continued through the 1930's. In 1942, the Depression ended with America's entry into World War II. The war accomplished what governmental programs could not by employing millions of Americans, either as soldiers or in the factories that manufactured war materials. The war put technological development into high gear, and many of the devices that have significantly affected our society were developed during this period. The post-war economy was booming when soldiers returned home, many to take factory jobs, marry and raise families. The impact of these developments on the central hardwood region was a shift in the lifestyle and objectives of land owners. Many people kept their residence in a rural setting while seeking employment in mills and factories elsewhere. The reason for owning land shifted away from utilitarian (production) objectives toward recreational/aesthetic enjoyment/ residential objectives (Birch and Kingsley 1978) (Figure 10).

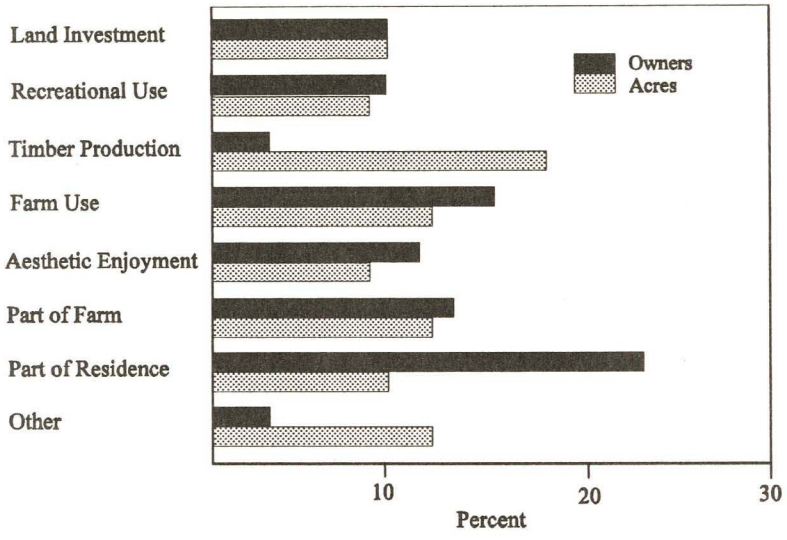



Figure 10. Reason for owning land in West Virginia (Birch and Kingsley 1978)

The changing land ownership characteristics have continued into the most recent decades throughout the central hardwood region. This is illustrated by the changes that occurred from 1957 to 1975 in West Virginia, a state that experienced little or no total population growth over that period. In 1957, the West Virginia Forest Industries Committee reported that there were 133,570 private owners of forestland, who owned over 8.9 million acres. The average size of ownership was 66.6 acres in 1957. By 1975, the number of owners had increased to 207,500 and the forestland acres had also increased to 10.3 million acres. But the average size of ownership had dropped to 49.8 acres (Birch and Kingsley 1978). Twenty percent of the land owners in 1975 were not from rural or farm backgrounds. Twenty-two percent of the land owners held their land primarily for aesthetic enjoyment or recreation, and another 23% owned the land primarily for their residence. Only 3% owned land primarily for timber production. Similar changes have taken place throughout the central hardwood region.

Alig (1990) reported the trends in forested land by states. If the 11 states that constitute the bulk of the central hardwood region are summarized (Table 3), the trend in total forestland is depicted in Figure 11. A decrease of about 2.5 percent in forestland is projected to occur by 2040, mostly due to urban expansion.

Table 3. Summary of forested lands by state in the central hardwood region

| States | (Millions Acres) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | Projected | | | | |
| | 1952 | 1962 | 1970 | 1977 | 1987 | 2000 | 2010 | 2020 | 2030 | 2040 |
| NC | 19.6 | 20.0 | 20.1 | 19.4 | 18.4 | 17.7 | 17.6 | 17.5 | 17.5 | 17.5 |
| VA | 15.5 | 15.7 | 15.9 | 15.9 | 15.4 | 15.0 | 15.0 | 15.0 | 15.0 | 15.0 |
| AK | 19.6 | 19.9 | 18.0 | 16.8 | 16.6 | 15.6 | 15.4 | 15.1 | 15.1 | 15.1 |
| KY | 11.5 | 11.6 | 11.8 | 11.9 | 11.9 | 12.1 | 12.2 | 12.3 | 12.5 | 12.5 |
| TN | 12.5 | 13.4 | 12.8 | 12.9 | 12.6 | 12.4 | 12.3 | 12.3 | 12.3 | 12.3 |
| MD | 2.9 | 2.8 | 2.7 | 2.5 | 2.5 | 2.4 | 2.3 | 2.2 | 2.1 | 2.1 |
| MA | 3.3 | 3.0 | 2.8 | 2.8 | 3.0 | 3.0 | 2.9 | 2.8 | 2.7 | 2.6 |
| PA | 14.6 | 16.3 | 16.1 | 15.9 | 16.2 | 16.3 | 16.3 | 16.0 | 16.0 | 15.8 |
| WV | 10.3 | 11.4 | 11.4 | 11.5 | 11.8 | 11.8 | 11.8 | 11.8 | 11.8 | 11.8 |
| OH | 5.4 | 6.0 | 6.4 | 6.9 | 7.1 | 7.3 | 7.3 | 7.1 | 7.0 | 7.0 |
| MD | 14.3 | 13.5 | 12.5 | 12.3 | 12.0 | 12.0 | 12.0 | 11.9 | 11.9 | 12.0 |
| Total | 129.5 | 133.6 | 130.5 | 128.8 | 127.5 | 125.6 | 125.1 | 124.0 | 123.9 | 123.7 |
| Change (from 1987) | +2.0 | +6.1 | +3.0 | +1.3 | 0 | -1.9 | -2.4 | -3.5 | -3.6 | -3.8 |
| Percent Change | +1.6% | +4.8% | +2.3% | +1.0% | 0 | -1.5% | -1.9% | -2.7% | -2.8% | -2.9% |

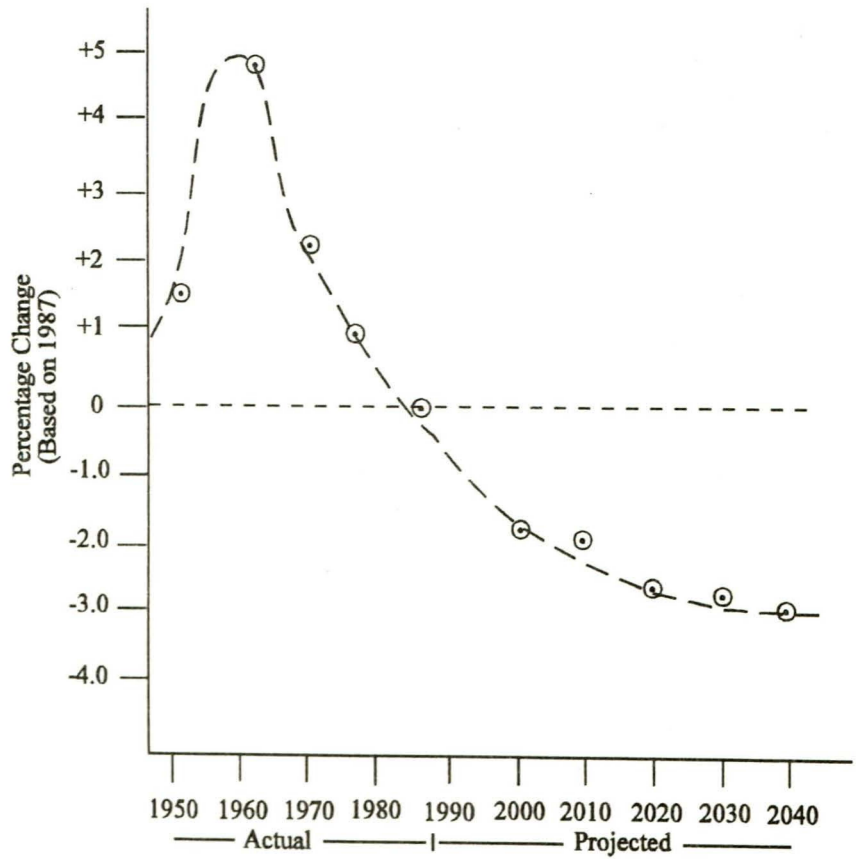



Figure 11. Change in forestland (actual and predicted) for 11 states with significant area within the central hardwood region (AR, KY, MA, MD, MO, NC, OH, PA, TN, VA WV), 1950-2040, from Alig (1990).

Agricultural policy continued to have an impact on forestry throughout the post-war period. The Conservation Reserve Program in the 1960's provided compensation to farmers to convert cropland to forest by planting trees. The Forest Incentives Program provide cost sharing to small private land owners for tree planting and other forest practices. The Stewardship Incentives Program of the 1990's provides federal cost sharing to small forest land owners to develop and carry out planned forest management. Some states have offered additional property tax incentives to land owners enrolled in the Stewardship Incentives Program. These programs are designed to promote good forest management among the many and varied small private land owners, a type of ownership that predominates in the central hardwood region. Technical support for forestry is provided to private land owners through state forestry agencies, Agriculture Extension Agents and the USDA Forest Service, State and Private Forestry. But in spite of this large effort, the level of participation among private forest land owners in the central hardwood region has generally lagged behind government expectations, partly due to the inherent independence of property owners, partly due to a lack of interest or economic incentives and partly due to an inability to inform land owners of the availability of such programs.

The withdrawal of forestland for agriculture leveled off in about 1933, but during the post-war decades, some decline in forest area has occurred. This has mostly resulted from development-related activities. One example is the interstate highway system which was developed during the 1960's and 1970's for the most part. Rights-of-way for gas transmission and electric power transmission also consume large corridors through the forest, and surface mining and urban development have taken up significant amounts of forestland. Topography and economic factors have resulted in the central hardwood region's being an area that is currently among the highest in the eastern United States for coverage with natural vegetation (Kolpatek and others 1979) (Figure 12).

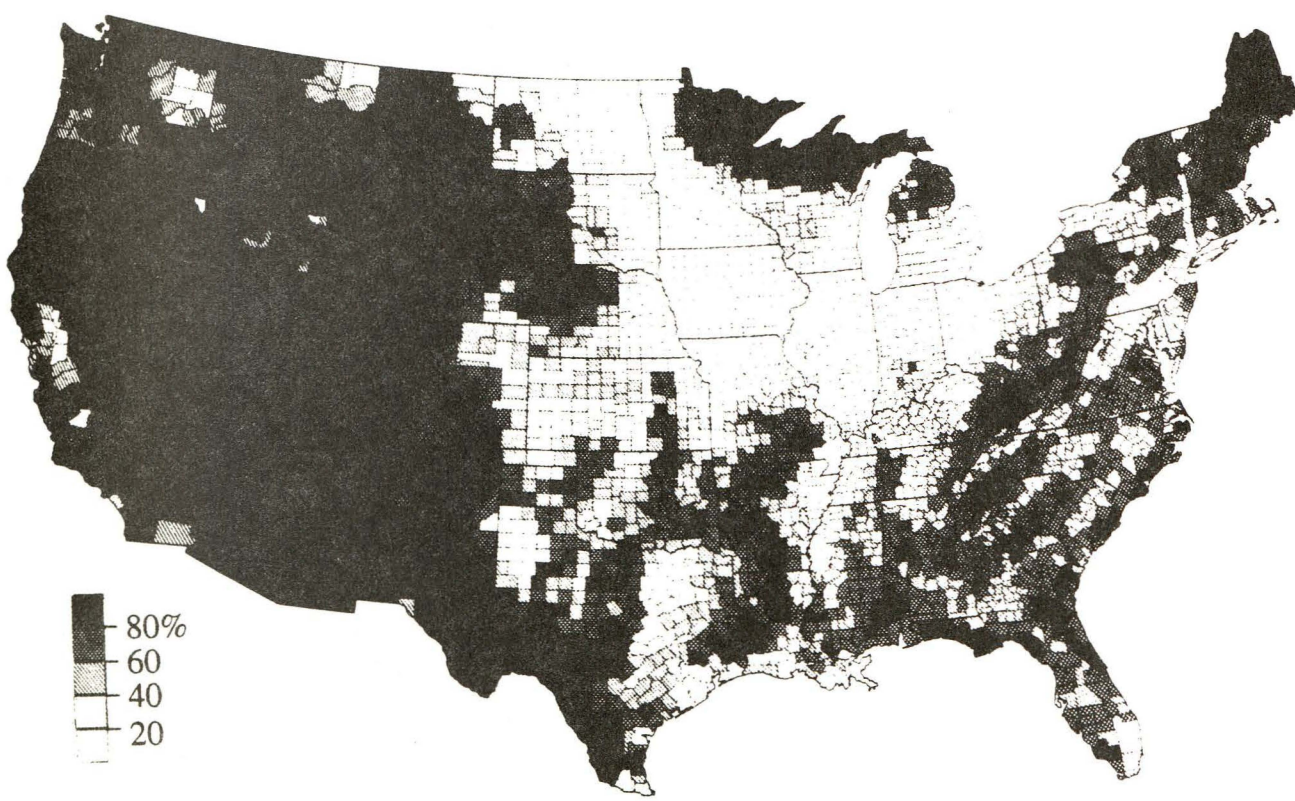



Figure 12. Coverage of natural vegetation by county in the US (Klopatek and others 1979).

Some of the most significant changes to the central hardwood forest in recent years have been brought about as a result of human introduction of insects and disease organisms. These changes are often not dramatic, but their impact is unquestionably large. The list of important introduced pests includes dogwood anthracnose, Dutch elm disease, oak wilt and beech bark disease. But the two with the most impact or potential impact on the central hardwood forest are **chestnut blight** and **gypsy moth**. The former is a disease that has effectively eliminated a species, which at one time was among the most widely distributed and economically important in the central hardwood region. The latter is an introduced insect that continues to spread southwestward through the central hardwood region and seems to have the potential to spread through the entire region over the next 50 to 75 years.

The **chestnut blight** fungus (*Cryphonectria parasitica*) was introduced to the New York Botanical Gardens in 1904 (Murrill 1904, Giddings 1912). By 1915 most states east of the Mississippi River reported infected trees. By the late 1930's, most mature chestnut trees were dead throughout the eastern states. The following should provide a frame of reference for the significance of American chestnut. Chestnuts were a source of food and commercial enterprise for mountain farmers. During the settlement period, log cabins were often constructed of chestnut logs due to their durability and decay resistance. Most split-fence rails and posts were constructed of chestnut where it was available. And due to its durability, chestnut lumber was widely used in construction of barns, sheds and even for the manufacture of furniture. The bark of chestnut was an important source of tannin for leather tanning factories.

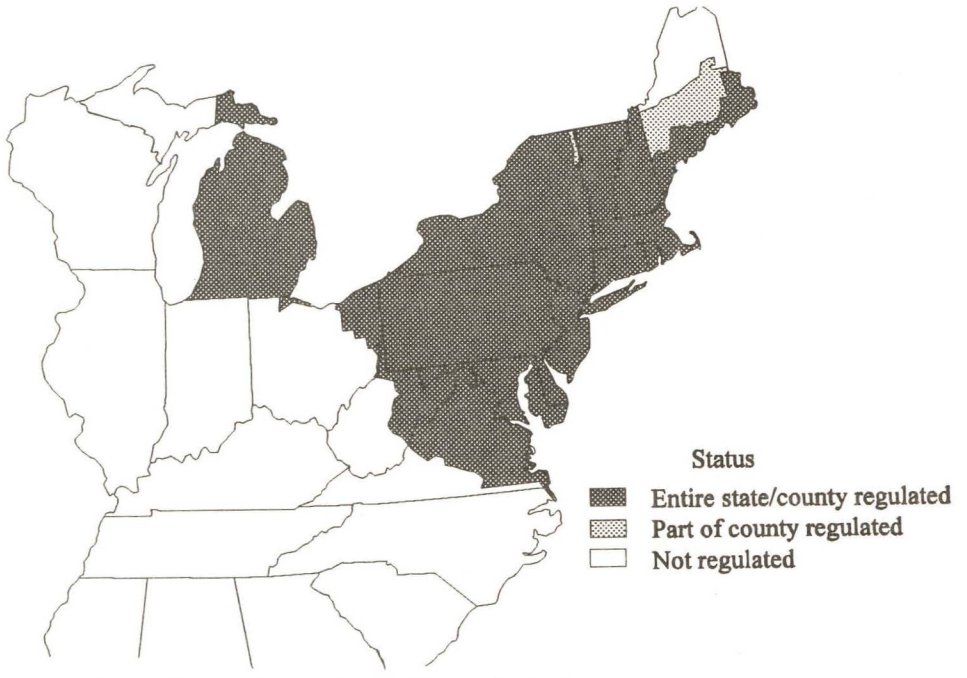
Chestnut also occupied an important position in the hardwood ecosystem. Braun (1950) used the name **oak-chestnut** to identify one of the most significant forest formations in the eastern deciduous forest. American chestnut was an important source of mast for a variety of wildlife species (animals and birds) most of which served as prey for larger predatory mammals. The impact of the loss of American chestnut can never be fully assessed. But Brooks (1915) indicated the following facts regarding chestnut in West Virginia alone. The annual harvest of chestnut in West Virginia in 1915 was 118 million board feet. The entire volume of chestnut in West Virginia was estimated to be 5 billion board feet. At a stumpage value of $8 per thousand board feet, in 1915, the entire chestnut inventory was estimated to be worth $55 million. At current stumpage prices for red oak that value would be about $1.5 billion. The only evidence remaining of the once important chestnut are occasional root sprouts, which usually survive to the sapling stage before being killed by the blight, and the grey remains of stumps of salvaged trees. Ironically, many of these stumps still have coatings of charcoal, evidence of the fires that burned over the area, probably after the chestnuts had already succumbed to the blight.

**Gypsy moth** (*Lymantria dispar*) is an introduced insect that potentially could have as great, or greater, impact on the central hardwoods than chestnut blight. Gypsy moth was inadvertently introduced into the Boston area in 1869 by Leopold Trouvelot (Liebhold 1989), who was interested in hybridizing the gypsy moth with native silkworms. The spread of gypsy moth has continued, and it is potentially capable of becoming established throughout the central hardwood region. Although gypsy moth can defoliate a variety of tree species, including conifers, deciduous species are their preferred hosts. Oaks are among the most preferred hosts of gypsy moth (Bess and others 1947, Gansner and Herrick 1987). Black cherry, hickory and maples are intermediate in preference while species, such as yellow-poplar and ashes, are virtually resistant. The significance of this to the central hardwood region is the fact that susceptible species make up the bulk of the forest composition throughout the region, the exception being yellow-poplar, especially in the mixed mesophytic forest.

Currently, gypsy moth has become established in, and is endemic to, an area stretching from Michigan to North Carolina (Figure 13). But isolated populations occur throughout the central hardwood region, having been spread by movement of vehicles, logs or other objects containing egg masses.

As a defoliator, gypsy moth's impact on the tree is to reduce its photosynthetic surface. The major defoliation occurs during the spring (May-June) which is a period of critical importance to deciduous trees since the soil moisture is generally high and the temperature is moderate during this time. Usually one defoliation is not lethal to trees, but a single defoliation during a drought year or multiple years of defoliation can lead to extensive tree mortality (Fosbroke and Hicks 1989). The factors that precipitate extensive gypsy moth outbreaks include presence of a suitable host plus climatic conditions that favor buildup. Since gypsy moth is situated in an oak-dominated region, the host is generally suitable, thus outbreaks can occur whenever weather triggers them. As can be seen in

Figure 14, cycles of outbreaks seem to occur about every 10 years. The outbreaks in recent years have been affecting more acres than in previous outbreaks, mostly due to the spread of gypsy moth to occupy a larger endemic range.




Map provided by the USDA Animal and Plant Health Inspection Service

Figure 13. Gypsy moth quarantine area as of 1996.

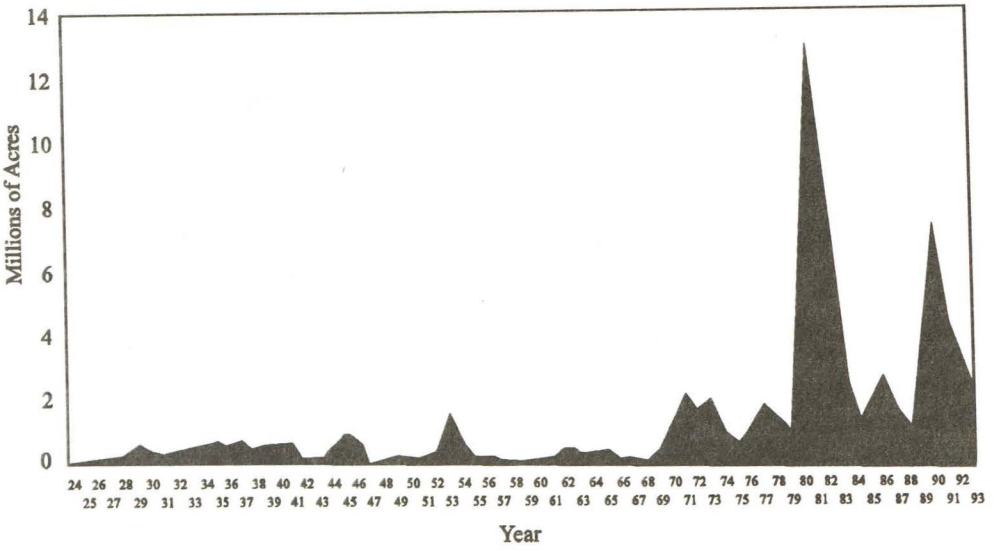



Figure 14. Outbreak history of gypsy moth in the Northeast.

The gypsy moth story is still unfolding, but its impact on the central hardwood forest has already been experienced throughout the portion of the region where it has spread. As reported by Gansner and Herrick (1987), oak experienced an average mortality rate of 24.2% between 1979 and 1984. Quimby (1987) reported a similar result where summer droughts plus gypsy moth defoliation in 1980-1983 resulted in the premature loss of 68 million trees in Pennsylvania, a mortality rate of 27.6%. Much of the mortality occurred in oak stands. Hicks and Fosbroke (1987) reported similar rates of tree mortality in southwestern Pennsylvania and western Maryland. The forest invaded by gypsy moth as it entered northeastern Pennsylvania has been dubbed "the new frontier" by Herrick and Gansner (1988), and essentially this new frontier represents the oak-dominated forest of the central hardwood region. The reduction in oak stocking that has occurred in the part of the central hardwood forest where gypsy moth has become established will most likely represent what can happen throughout the entire region.

Another episode in recent history that has had a significant impact on the central hardwood forest is the decline and later reestablishment of **wildlife populations** across the region. In addition to white-tailed deer, elk, and bison were found extensively throughout the region prior to European settlement. Those species were, in part, hunted into extinction but mostly were displaced due to habitat destruction by land clearing for farming. Farmers also eradicated predator species, such as eastern mountain lion, eastern timber wolf and black bear, since these predators were a threat to their livestock. By 1910, bison, elk, mountain lion, and wolf had been eradicated from the central hardwood forest, and white-tailed deer and black bear were found only in small isolated populations in remote and mountainous regions.

Wildlife advocates and hunters pushed for, and obtained, stricter hunting regulations, such as seasonal hunting, bag limits and bucks-only hunting. The results for white-tailed deer have been remarkable. In the absence of natural predators, and with the second-growth hardwood forest providing prime habitat, the deer herd has risen sharply throughout the central hardwood region. In fact, deer have generally become a serious problem to forest regeneration in many areas. Deer consume 1-1.5 kg of vegetation daily, consisting of twigs, acorns and some grass and herbs (Smith 1993). Deer are selective and prefer seedlings of oaks and maples to those of black cherry (Marquis 1981). Overbrowsing by deer can result in development of a fern-dominated understory where tree seedlings have a difficult time getting established due to competition and allelopathy (plant toxins). Thus, the long-term effect of deer on the forest may be to retard the process of regeneration, selectively change the species composition and reduce the overall diversity of species.

The modern chapter in forest history also includes a marked change in the **social context** of forest resources. This has been brought about by a combination of factors including the changing demographics in America, where people today are physically removed from the land. But other factors have come into play. For example, abusive and exploitive land uses, such as exploitive timbering, unregulated mineral extraction, air and water pollution, and uncontrolled development are all sources of public concern. The environmental movement began to take shape as a cultural phenomenon during the late 1960's. Rachel Carson's book, *Silent Spring*, did for environmentalism what Harriett Beecher Stowe's *Uncle Tom's Cabin* did for the emancipation movement. Against a backdrop of Kennedy altruism and polarizing issues, such as the cold war, racism and a budding Viet Nam conflict, it seems, in retrospect, inevitable that the environmental movement, led by altruistic urban youth, should flourish. Most foresters, at that time, were from rural backgrounds and had come through their forestry training with a paternalistic approach toward resource management, being one in which the forester is regarded as the trained professional, and should be trusted to make the appropriate decisions regarding management of the resource. Thus, the elements of conflict were in place. The political arena became the venue for the struggle. Environmental activist groups gained strength. The **Wilderness Act** of 1964 represented a milestone in environmental policy legislation whereby areas in the National Forests could be set aside for "wilderness," and timber cutting in these areas would be prohibited (Dana and Fairfax 1980). Controversy over the use of clearcutting on public land resulted in the landmark **Monongahela Decision** of 1970 that recommended uneven-age management as a primary management policy for the Monongahela National Forest in West Virginia. This decision was more significant as a precedent than it was to the Monongahela National Forest. In effect, it imposed the will of the public over that of foresters, who had always been regarded as the experts in forest resource management.

The issue of the 90's seems to revolve around regulation of private land. The so-called "spotted owl" controversy in the Pacific Northwest has set the tone. The perceived situation was one in which an endangered organism's habitat (old-growth forest) was not limited to a single land owner but extended to private land as well as public land.

The debate extending to small non-industrial private forest (NIPF) owners has progressed from the passage of regulations requiring the use of "best management practices" for logging to a more comprehensive and controversial debate over whether or not land use should be controlled at the **landscape level**. If such control is needed to facilitate what is called **ecosystem management**, how can it work in areas, such as the central hardwood region, where over 75% of the land is owned by NIPF owners? Proposed solutions include a combination of government educational and assistance programs plus incentive programs (Campbell and Kittredge 1996). The use of legislative regulation to achieve the goals of ecosystem management is a controversial approach (Argow 1996) and one that evokes strong emotions. The fear among private owners—that they will lose prerogatives to use their land—is not unfounded. Examples abound where the "general good" of the public has taken precedence over the desires of the individual.

On the other side of the issue are those representing private land owners (Farm Bureau, National Woodland Owners Association, etc.) who contend that the loss of management options to a private land owner due to regulation constitutes a "taking" under the Fifth Amendment of the Constitution, for which the owner is entitled to be compensated. As these debates continue, the outcomes will most certainly affect the way forests can be, and will be, managed and exploited in the central hardwood region in the years to come.

In summary, the history of the central hardwood forests is one that is based on the actions and interactions of climatic and biological factors, and, in more recent times it has been strongly affected by humans and their actions (Table 1). Going back to the retreat of the last glaciers (12,000-14,000 years ago) and the climatic warming trend that has followed, the deciduous forests found an ideal environment and flourished in the central hardwood region. Indigenous people used fire and cultivated land, and their activities peaked about the time European settlement began. The forest regrew for about 200 years, and then clearing for subsistence farming began in ernest. This proceeded until about the time of the Civil War and the onset of the industrial revolution. This period marked the beginning of a shift in agriculture to more level land, and abandoned hill lands in the central hardwood region began to regrow into forest. A period of exploitive logging around the turn of the 20th century has been followed by a regrowing forest. Forest land owners and the public have changed in their attitudes concerning forest resources in recent times. Now with many hardwood stands in the central hardwood region approaching 100 years of age, the resource is maturing. Many forces are mustering for a piece of the action. Developers, speculators, industries, agencies, advocacy groups and environmentalists are all becoming aware of this unique and diverse forest. Our actions today will be the history of the future, and foresters should be prepared to participate in, and hopefully guide, the decision-making process at this important crossroad in the central hardwood regions development.

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# SUSTAINING BIODIVERSITY IN MIDWESTERN WOODLANDS

Douglas Ladd[1]

**Abstract:** Woodland ecosystems in the Midwestern United States provide habitat for an impressive array of biodiversity, including endemic organisms. Sustainable conservation of high quality woodland landscapes must consider the genesis, process regimes, and ecological dynamics of these systems, both in contemporary and presettlement time frames. An essential goal of conservation efforts must be to retain diverse aggregations of native biota. To this end, organismal considerations are requisite for implementing and monitoring woodland ecosystem management from a biodiversity standpoint. Examples of woodland restoration and management are presented and discussed, with particular emphasis on fire ecology and biodiversity impacts.

[1]The Nature Conservancy, 2800 S. Brentwood Blvd., St. Louis, MO 63144.

# INDUSTRY'S NEEDS AND CONCERNS

William Luppold[1]

Abstract: The central hardwood resource is utilized by a diverse group of industries with varying timber requirements. Most saw and veneer mills desire high quality sawtimber of a few select species. Over the last 20 years, stumpage prices of these species have increased faster than lumber prices as a result of competitive markets and an apparent shortage of high-grade sawtimber. Furthermore, reduced supplies of high-grade timber on private lands are occurring at a time when National Forest timber is becoming less accessible. In the short run, lumber and veneer industries have been able to absorb increased timber prices by better production and marketing practices. However, it is feared that eventually the lack of high-quality sawtimber will require building or retooling production facilities to better utilize lower grade material. The pulp and composite industry has become a major user of hardwood timber during the last two decades. Because these operations are capital intense and technologically complex, production can only be stopped at a high cost. This makes continuous material supplies critical. The long-term concern of the pulp industry is not having adequate volumes of material priced low enough to be competitive in an international market. Perhaps the most important emerging issue is whether the different users of hardwood sawtimber will cooperate with or compete with one another. Although the pulp and composite industries consume low-grade material, the sheer growth of these industries has caused the sawmill industry to be concerned over long-term timber supply. Still, demand by pulp and composite industries is allowing for greater utilization of timber on poorer sites and in some instances can actually increase the volume of higher grade material available. These markets for low-grade material also may allow for thinning options that have been historically uneconomical. Although nothing is certain in hardwood markets, the industries that derive the most value from the forest resource will be able to pay the highest price for the timber resource.

[1]Project Leader, USDA Forest Service, Northeastern Experiment Station, Princeton, West Virginia.

# INTEGRATING SILVICULTURE, FOREST MANAGEMENT, AND FOREST POLICY

Chadwick D. Oliver[1]

Abstract: Much progress has been made in developing and implementing individual silvicultural operations such as regeneration, thinning, and harvesting by various means. Similarly, foresters have made progress at managing flows of timber and other commodity values. Present concern that forests provide an even greater variety of commodity and non-commodity values can be accommodated by integrating management at the various levels—silvicultural operations, silvicultural regimes, landscape patterns, forest management, and policy. Using modern, systems management approaches, the integration can be done without the inefficiencies of central planning, regulations, or a "command-and control" approach. Especially important are the landscape and policy levels. Management at the landscape level will generally attempt to maintain a diversity of stand structures across the landscape; therefore, a variety of silvicultural operations and regimes will be done to different stands. Computerized tools are allowing management at this level to be done very effectively, by coordinating the stands and treating the landscape as a "portfolio" in which the forest in each stand is an "asset" which matures at a specific time for a specific market, this landscape management can actually be more profitable. Policies to encourage different landowners—small and large, private and public—to provide the many commodity and non-commodity values at the different levels can be developed. The values can best be achieved using incentive-based policies without imposing restrictions on individual landowners. The result of such organization and policies can be a greater provision of nearly all values to society and greater profit to the landowners.

[1]College of Forest Resources, University of Washington, Seattle, WA 98195.

# EVALUATING TIMBER HARVESTING IMPACTS ON WILDLIFE HABITAT SUITABILITY USING FOREX

Chris B. LeDoux[1]

Abstract: Precommercial, commercial, and final harvesting operations can impact wildlife habitat suitability by altering the vegetation composition on a given site. Harvesting operations remove trees and many times provide the necessary perturbation to trigger successional conditions different from those that existed prior to the harvest. Although these new successional changes can be beneficial to a wide array of wildlife species, they also can alter the site such that some wildlife species that were once abundant will no longer find the site suitable. In this paper, FOREX is used to demonstrate how managers, loggers, and planners can evaluate the impact of proposed harvest treatments on wildlife habitat suitability. FOREX is an integrated expert system that explicitly considers the potential growth and yield of a stand, wood products and their development over time, the logging technology needed for harvests scheduled, the economic and market factors involved, and the impacts on wildlife habitat. Initial results suggest that a series of light thinnings can provide positive cash flows to the landowner and meet the habitat needs for many species of birds, amphibians, reptiles, and mammals for a long period of time. The results also show that removing all the downed, dead woody material from the forest floor is detrimental to many species of wildlife, and that the final harvest has the most impact on wildlife habitat suitability.

## INTRODUCTION

As managers of forests and woodlots in the hardwood region of the Eastern United States try to meet increasing world demand for wood products, they are faced with numerous challenges. This demand is not only for wood products but also for suitable habitat for wildlife, visual and aesthetic quality, clean water, recreation, and social values. A major challenge for owners of forested land is to meet both financial and personal objectives while achieving goals at the ecosystem or landscape level.

These same forests are under occasional stress from several agents such as the gypsy moth, beech bark disease, forest tent caterpillar, pear thrips, and elm spanworm (Twardus 1995). A major threat to the present day oak-hickory forests is the gypsy moth. Hickory (*Carya* spp.) has declined due to several wood-boring insects, whereas white pine (*Pinus strobus*), which is associated with many different hardwood species, is coming back. White ash (*Fraxinus americana*) and black ash (*Fraxinus nigra*) are also declining due to a disease, ash yellows, caused by a mycoplasma-like organism. Ash anthracnose, a foliage disease, is also contributing to severe dieback and poor crown density. Butternut (*Juglans cinerea*), a highly valued tree for wildlife mast production and lumber products, is threatened by a devastating disease known as butternut canker (Twardus 1995).

Some of these same forest species are also experiencing serious regeneration problems because of high deer populations in some areas (Stout and others 1995). The change in harvesting practices from clearcutting to partial harvest methods such as single-tree, group selection, and selection cuts have added to the regeneration challenges (Sander and Clark 1971, Dale and others 1995, Weigel and Parker 1995). These contemporary harvest methods, although more costly to apply and less efficient for timber production, are being used in response to increasing concerns of the public to save and protect the existing forests (LeDoux and others 1991, 1993, 1994). The commitment to timber production in combination with ecosystem or landscape-level goals further complicates the task of managing forested lands.

[1]Supervisory Industrial Engineer, USDA Forest Service, 180 Canfield Street, Morgantown, WV 26505-3101.

Of particular interest to all is how specific management activities affect the suitability of wildlife habitat. This challenge is further complicated by a lack of decisionmaking tools for integrated analysis. Accordingly, we have developed an expert system called FOREX that allows for integrated decisionmaking in the management of hardwoods forest (LeDoux and others 1995, 1996). FOREX considers the potential growth and yield, products and development of a stand over time, economic and market factors, and impacts on wildlife habitat. This system can be applied to all forest types in the Northeast, and currently considers impacts on wildlife habitat for northern red oak and oak-hickory forest types. As wildlife data on other forest types and regions become available, they will be incorporated into the FOREX database. In this example, FOREX is applied to hypothetical harvesting situations that a landowner might consider to assess the management impacts on wildlife habitat suitability.

## METHODS

### Description of FOREX and MANAGE

FOREX and MANAGE were the models used in this analysis. FOREX is an integrated expert system that uses data from simulation runs from MANAGE (LeDoux 1986; LeDoux and others 1995, 1996). MANAGE, a computer program written in FORTRAN V, integrates harvesting technology, silvicultural treatments, market price, and economic concerns over the life of a stand. The simulation is a combination of discrete and stochastic subroutines. Individual subroutines model harvesting activities, silvicultural treatments, growth projections, market prices, and discounted present net worth (PNW) economic analysis. Specifically, the model allows the manager to evaluate how alternative harvesting technology, silvicultural treatments, market price, and economic combinations affect costs and benefits over the life of a stand. The model uses a detailed individual user-specified tree list and then projects stand growth based on some user-specified silvicultural treatment, harvests the desired volume or stems with the logging system specified, sells the wood, and conducts economic analysis for the respective treatment and entry. MANAGE was run for numerous stands in the northern red oak and oak-hickory forest type with alternative combinations of silvicultural treatment and logging technology. The results from these runs were stored in the FOREX database. Only results from runs that breakeven or better were stored in the database.

Using FOREX, the user can obtain information on PNW, optimal thinning entry timing, optimal stand rotation age, diameter at breast height (dbh), volume by grade and value of the trees harvested, and, based on the logging system used, average slope yarding distance, truck class, road class, log-bucking methods, and number of thinnings desired. FOREX also provides information on the effect of harvesting treatments on wildlife habitat suitability. The user can obtain information on the PNW, dbh, and volume required for a specific set of management objectives, and perform a sensitivity analysis that eliminates the need to sort through numerous simulations.

### Harvesting Treatments

The harvest treatments evaluated include precommerical thinning, multiple commercial thinnings, and final harvest. The objective for each treatment considered is to grow quality wood products, and in all cases except the precommercial thinning, to breakeven or return a profit to the landowner. The impact on wildlife habitat suitability is evaluated by focusing on the number of wildlife species for which the site would be suitable. Although species composition based on densities may change with harvesting treatments, we specifically focus on the potential numbers of amphibians, birds, reptiles, and mammal species. In some cases, woody debris in the form of cull logs and tops is left on the site and contrasted with removing all cull logs and tops.

### Site and Stand Conditions

In this example, the land holding is a 300-acre forest in the northern red oak cover type. The species mix includes red oak (*Quercus rubra*), red maple (*Acer rubrum*), hickories (*Carya spp.*), black oak (*Quercus velutina*), scarlet oak (*Quercus coccinea*), and chestnut oak (*Quercus prinus*). The average site index of the stand is about 80. The stand is 30 years old and contains 326 trees per acre that are more than 5 inches dbh. The average dbh is 6.15 inches. The land is assumed to be located on gentle to moderate slopes and requires ground-based systems for harvests. The site is covered with greater than 25% rock cover, has greater than 30% forest litter cover, has active seeps, loose soils

and small caves, and has at least 50 $ft^3$ per acre of dead and down material on the forest floor. The land has about 20% shrub cover and at least 29% ground cover. The forest model plot chosen for this analysis is representative of the northern red oak forest type.[2] The landowner wants to evaluate the impact of alternative harvest treatments and wood utilization options on wildlife habitat suitability.

## RESULTS

The landowner's property attributes and objectives were incorporated into FOREX, which sorts through the database for combinations of conditions that will meet the user's objectives. The program returns the results for the conditions that will meet all objectives. The results for each harvesting treatment evaluated in this analysis are shown in Table 1.

Table 1. FOREX results by treatment

| Attribute | Precommercial thinning[1] | First commercial thinning[2] | Second commercial thinning[2] | Final harvest |
|---|---|---|---|---|
| Yarding distance (ft) | 800 | 800 | 800 | 800 |
| Buck type | 1 | 1 | 1 | 1 |
| Road class | 3 | 3 | 3 | 3 |
| Truck class | 2 | 2 | 2 | 2 |
| Age (years) | 60 | 90 | 150 | 160 |
| Trees (no.) | 28 | 70 | 34 | 74 |
| Dbh (inches) | 9.93 | 12.32 | 17.97 | 18.98 |
| Volume ($ft^3$) | 375.60 | 1503.37 | 1739.67 | 4369.99 |
| G1 (bd. ft.)[3] | 0 | 693.79 | 3965.23 | 1437.03 |
| G2 (bd. ft.)[4] | 0 | 0 | 787.65 | 0 |
| G3 (bd. ft.)[5] | 23.34 | 536.06 | 448.34 | 673.93 |
| G4 ($ft^3$)[6] | 368.67 | 1156.97 | 714.92 | 1562.08 |
| PNW ($) | -121.47 | 45.95 | 51.30 | 127.91 |
| Cash flow ($) | -294.85 | 270.73 | 1780.68 | 5966.89 |

[1] 10% of trees/acre removed.
[2] 30% of trees/acre removed.
[3] G1 = grade 1 volume.
[4] G2 = grade 2 volume.
[5] G3 = grade 3 volume.
[6] G4 = pulpwood.

The precommercial thinning would remove about 10% of the existing trees/acre and would be scheduled for age 60. The average dbh of material removed is 9.93 inches and about 375.60 $ft^3$ of wood products are harvested per acre. The wood removed is all pulpwood since the trees are too small to make sawlogs. This treatment could be prohibitively expensive and would likely never be implemented.

[2]U.S. Department of Agriculture, Forest Service. One-acre forest model plot. Unpublished data on file at Morgantown, WV: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station.

First and second thinnings were considered for meeting the multiple thinnings objective. They were considered alone. Precommercial thinning was not included in this set. The first and second commercial thinnings would remove 1,503.37 $ft^3$/acre and 1,739.67 $ft^3$/acre, respectively. The first commercial thinning at age 90 would return about $270.73/acre and the second thinning would return $1,780.68/acre. These thinnings become more profitable than precommerical thinning because we are removing larger trees that result in sawtimber products. The second thinning has a larger proportion of higher grade sawlogs.

The final harvest removes 4,369.99 $ft^3$ of wood products/acre and results in a cash flow of about $5,966.89 at age 160. The majority of the wood products are higher grade sawlogs. The trees measure about 18.98 inches dbh. About 1,562.08 $ft^3$ of pulpwood/acre is harvested at final harvest. This series of commercial thinnings and final harvest keep a stand of trees on the site for a period of about 130 years.

Integrating Wildlife Habitat Suitability

Forest wildlife populations and their habitats are products of the land and how it is managed for wood fiber (DeGraaf and others 1991, 1992) or in unmanaged stands from events that occur naturally. DeGraaf and others (1992) developed species/habitat matrices that forest managers, planners, silviculturists, loggers, and wildlife biologists can use to increase potential numbers of bird, amphibian, reptiles, and mammal species in New England forest types. Information is provided in the matrices for species occurrence and utilization by forested habitat and forest type by life history activities and seasons. These species/habitat matrices, which are tied to the seral states present in the development of a given forest type, have been incorporated into the FOREX database. With the FOREX program, actual stand attributes such as dbh, volume/acre, number of trees per acre, and species mix are matched with DeGraaf's and others (1992) guidelines to identify which species would find the site suitable by harvesting treatment. FOREX then reports the species of wildlife that would find the site suitable. The results are presented for before and after harvest conditions. The user can then gauge the impact of harvest treatments on wildlife habitat suitability.

The wildlife habitat suitability results from FOREX were validated by comparing them to results from the HAM model (Harvey and Finley 1995). The HAM model uses stand-level data, DeGraaf and others (1992) guidelines, and other wildlife data to predict wildlife species that may use the stand for breeding and feeding for the northern red oak and the oak-hickory forest types. Two levels of analysis are available. The first uses a small matrix constructed using cluster analysis to select functional groups of species that depend on one or more common habitat features. The second analysis uses an expert system, GNOSIS (Meyers and Foster unpublished[3]), developed at the Pennsylvania State University. GNOSIS uses the stand-level data to search the matrix species by species rather than by functional groups producing a list of species. Results from FOREX and HAM were identical for the conditions evaluated.

The stand and site conditions remain essentially the same before and after the precommerical, first, and second thinnings with respect to site attributes used in the guidelines (DeGraaf and others 1992). The proposed light thinnings do not alter habitat suitability because the residual stand conditions remain much the same as the initial stand.

Tables 2, 3, and 4 list the potential species that would find the site suitable before and after precommercial, first, and second thinnings.

[3]Meyers, W. and M. Foster. unpublished. GNOSIS: An expert system. Penn State Univ. Sch. of For. Res.

Table 2. Potential species for initial stand before and after precommercial thinning

| | |
|---|---|
| Redback salamander (*Plethodon cinereus*) | Blue jay (*Cyanocitta cristata*) |
| Northern brown snake (*Storeia d. dekayi*) | Common raven (*Corvus corax*) |
| Northern redbelly snake (*Storeia o. occipitomaculata*) | Ruffed grouse (*Bonasa umbellus*) |
| Northern ringneck snake (*Diadophis punctatus edwardsi*) | Red-eyed vireo (*Vireo olivaceus*) |
| Northern black racer (*Coluber c. constrictor*) | Scarlet tanager (*Piranga olivacea*) |
| Black rat snake (*Elaphe o. obsoleta*) | Purple finch (*Carpodacus purpureus*) |
| Eastern milk snake (*Lampropeltis t. triangulum*) | Masked shrew (*Sorex cinereus*) |
| Northern copperhead (*Agkistrodon contortrix mokeson*) | Northern red bat (*Lasiurus borealis*) |
| Timber rattlesnake (*Crotalus horridus*) | Coyote (*Canis latrans*) |
| Sharp-shinned hawk (*Accipiter striatus*) | Black bear (*Ursus americanus*) |
| Northern goshawk (*Accipiter gentilis*) | Red fox (*Vulpes vulpes*) |
| Great horned owl (*Bubo virginianus*) | White-tailed deer (*Odocoileus virginianus*) |
| Eastern wood-pewee (*Contopus virens*) | |

Table 3. Potential species for stand before and after first commercial thinning

| | |
|---|---|
| Redback salamander (*Plethodon cinereus*) | Eastern phoebe (*Sayornis phoebe*) |
| Northern brown snake (*Storeia d. dekayi*) | Blue jay (*Cyanocitta cristata*) |
| Northern redbelly snake (*Storeia o. occipitomaculata*) | American crow (*Corvus brachyrhynchos*) |
| Northern ringneck snake (*Diadophis punctatus edwardsi*) | Common raven (*Corvus corax*) |
| Northern black racer (*Coluber c. constrictor*) | Ruffed grouse (*Bonasa umbellus*) |
| Black rat snake (*Elaphe o. obsoleta*) | Red-eyed vireo (*Vireo olivaceus*) |
| Eastern milk snake (*Lampropeltis t. triangulum*) | Blackburnian warbler (*Dendroica fusca*) |
| Northern copperhead (*Akistrodon contortrix mokeson*) | Scarlet tanager (*Piranga olivacea*) |
| Timber rattlesnake (*Crotalus horridus*) | Purple finch (*Carpodacus purpureus*) |
| Sharp-shinned hawk (*Accipiter striatus*) | Masked shrew (*Sorex cinereus*) |
| Northern goshawk (*Accipiter gentilis*) | Northern red bat (*Lasiurus borealis*) |
| Red-shouldered hawk (*Buteo lineatus*) | Coyote (*Canis latrans*) |
| Broad-winged hawk (*Buteo platypterus*) | Black bear (*Ursus americanus*) |
| Great horned owl (*Bubo virginianus*) | Red fox (*Vulpes vulpes*) |
| Eastern wood-pewee (*Contopus virens*) | White-tailed deer (*Odocoileus virginianus*) |

Table 4. Potential species for stand before and after second commercial thinning

| | |
|---|---|
| Redback salamander (*Plethodon cinereus*) | Black-capped chickadee (*Parus atricapillus*) |
| Northern brown snake (*Storeia d. dekayi*) | Tufted titmouse (*Parus bicolor*) |
| Northern redbelly snake (*Storeia o. occipitomaculata*) | White-breasted nuthatch (*Sitta carolinensis*) |
| Northern ringneck snake (*Diadophis punctatus edwardsi*) | Brown creeper (*Certhia americana*) |
| Northern black racer (*Coluber c. constrictor*) | Red-eyed vireo (*Vireo olivaceus*) |
| Black rat snake (*Elaphe o. obosoleta*) | Blackburnian warbler (*Dendroica fusca*) |
| Eastern milk snake (*Lampropeltis t. triangulum*) | Scarlet tanager (*Piranga olivacea*) |
| Northern copperhead (*Agkistrodon contortrix mokeson*) | Brown-headed cowbird (*Molothrus ater*) |
| Timber rattlesnake (*Crotalus horridus*) | Purple finch (*Carpodacus purpureus*) |
| Sharp-shinned hawk (*Accipiter striatus*) | Carolina chickadee (*Parus Carolinensis*) |
| Northern goshawk (*Accipiter gentilis*) | Virginia opossum (*Didelphis virginiana*) |
| Red-shouldered hawk (*Buteo lineatus*) | Masked shrew (*Sorex cinereus*) |
| Broad-winged hawk (*Buteo platypterus*) | Little brown bat (*Myotis lucifugus*) |
| Red-tailed hawk (*Buteo jamaicensis*) | Silver-haired bat (*Lasionycteris noctivagans*) |
| Great horned owl (*Bubeo virginianus*) | Big-brown bat (*Eptesicus fuscus*) |
| Red-bellied woodpecker (*Melanerpes carolinus*) | Northern red bat (*Lasiurus borealis*) |
| Yellow-bellied sapsucker (*Sphyrapicus varius*) | Gray squirrel (*Sciurus carolinensis*) |
| Hairy woodpecker (*Picoides villosus*) | Southern flying squirrel (*Glaucomys volans*) |
| Eastern wood-pewee (*Contopus virens*) | Northern flying squirrel (*Glaucomys sabrinus*) |
| Eastern phoebe (*Sayornis phoebe*) | Coyote (*Canis latrans*) |
| Blue jay (*Cyanocitta cristata*) | Black bear (*Ursus americanus*) |
| American crow (*Corvus brachyrhynchos*) | Red fox (*Vulpes vulpes*) |
| Common raven (*Corvus corax*) | White-tailed deer (*Odocoileus virginianus*) |
| Ruffed grouse (*Bonasa umbellus*) | |

Table 5 lists the species that would find the site suitable in the stand prior to final harvest. Table 6 lists the species that would find the site suitable after final harvest if at least 50 $ft^3$ of down wood and logs/tops were left on the site. Table 7 lists the potential species that would find the site suitable after final harvest if all down wood and logs/tops were removed from the stand. The differences in potential species numbers at each seral state are due to changes in habitat structure (Smith 1962). The largest potential species abundance and diversity is in the 90- to 160-year-old stands. Figure 1 shows the number of amphibians, birds, reptiles, and mammals by stand age for before and after treatment conditions. The potential species before and after harvest conditions remain the same for precommercial and commercial thinnings because we are removing 10 to 30% of the stand, leaving conditions essentially the same as prior to harvest.

Table 5. Potential species for stand before final harvest.

| | |
|---|---|
| Redback salamander (*Plethodon cinereus*) | Black-capped chickadee (*Parus atricapillus*) |
| Northern brown snake (*Storeia d. dekayi*) | Tufted titmouse (*Parus bicolor*) |
| Northern redbelly snake (*Storeia o. occipitomaculata*) | White-breasted nuthatch (*Sitta carolinensis*) |
| Northern ringneck snake (*Diadophis punctatus edwardsi*) | Brown creeper (*Certhia americana*) |
| Northern black racer (*Coluber c. constrictor*) | Red-eyed vireo (*Vireo olivaceus*) |
| Black rat snake (*Elaphe o. obosoleta*) | Blackburnian warbler (*Dendroica fusca*) |
| Eastern milk snake (*Lampropeltis t. triangulum*) | Scarlet tanager (*Piranga olivacea*) |
| Northern copperhead (*Agkistrodon contortrix mokeson*) | Brown-headed cowbird (*Molothrus ater*) |
| Timber rattlesnake (*Crotalus horridus*) | Purple finch (*Carpodacus purpureus*) |
| Sharp-shinned hawk (*Accipiter striatus*) | Carolina chickadee (*Parus carolinensis*) |
| Northern goshawk (*Accipiter gentilis*) | Virginia opossum (*Didelphis virginiana*) |
| Red-shouldered hawk (*Buteo lineatus*) | Masked shrew (*Sorex cinereus*) |
| Broad-winged hawk (*Buteo platypterus*) | Little brown bat (*Myotis lucifugus*) |
| Red-tailed hawk (*Buteo jamaicensis*) | Silver-haired bat (*Lasionycteris noctivagans*) |
| Great horned owl (*Bubeo virginianus*) | Big-brown bat (*Eptesicus fuscus*) |
| Red-bellied woodpecker (*Melanerpes carolinus*) | Northern red bat (*Lasiurus borealis*) |
| Yellow-bellied sapsucker (*Sphyrapicus varius*) | Gray squirrel (*Sciurus carolinensis*) |
| Hairy woodpecker (*Picoides villosus*) | Southern flying squirrel (*Glaucomys volans*) |
| Eastern wood-pewee (*Contopus virens*) | Northern flying squirrel (*Glaucomys sabrinus*) |
| Eastern phoebe (*Sayornis phoebe*) | Coyote (*Canis latrans*) |
| Blue jay (*Cyanocitta cristata*) | Black bear (*Ursus americanus*) |
| American crow (*Corvus brachyrhynchos*) | Red fox (*Vulpes vulpes*) |
| Common raven (*Corvus corax*) | White-tailed deer (*Odocoileus virginianus*) |
| Ruffed grouse (*Bonasa umbellus*) | |

Table 6. Potential species for stand after final harvest with down logs and tops left on site

| | |
|---|---|
| Northern brown snake (*Storeia d. dekayi*) | Common raven (*Corvus corax*) |
| Northern ringneck snake (*Diadophis punctatus edwardsi*) | Ruffed grouse (*Bonasa umbellus*) |
| Northern black racer (*Coluber c. constrictor*) | Wild turkey (*Meleagris gallopavo*) |
| Black rat snake (*Elaphe o. obosoleta*) | Purple finch (*Carpodacus purpureus*) |
| Eastern milk snake (*Lampropeltis t. triangulum*) | Masked shrew (*Sorex cinereus*) |
| Timber rattlesnake (*Crotalus horridus*) | Coyote (*Canis latrans*) |
| Cooper's hawk (*Accipiter cooperii*) | Black bear (*Ursus americanus*) |
| Golden eagle (*Aquila chrysaetos*) | Red fox (*Vulpes vulpes*) |
| Blue jay (*Cyanocitta cristata*) | White-tailed deer (*Odocoileus virginianus*) |
| American crow (*Corvus brachyrhynchos*) | |

Table 7. Potential species for stand after final harvest with down logs and tops removed

| | |
|---|---|
| Northern ringneck snake (*Diadophis punctatus edwardsi*) | Ruffed grouse (*Bonasa umbellus*) |
| Timber rattlesnake (*Crotalus horridus*) | Wild turkey (*Meleagris gallopavo*) |
| Cooper's hawk (*Accipiter cooperii*) | Purple finch (*Carpodacus purpureus*) |
| Golden eagle (*Aquila chrysaetos*) | Coyote (*Canis latrans*) |
| Blue jay (*Cyanocitta cristata*) | Red fox (*Vulpes vulpes*) |
| American crow (*Corvus brachyrhynchos*) | White-tailed deer (*Odocoileus virginianus*) |
| Common raven (*Corvus corax*) | |

| Age | 30 | | 60 | | 90 | | 150 | | 160 | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Before | After | Before | After | Before | After | Before | After | Before | After |
| Amphibians | 0 | 0 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 0 |
| Birds | 7 | 7 | 10 | 10 | 15 | 15 | 25 | 25 | 25 | 8 |
| Mammals | 3 | 3 | 6 | 6 | 6 | 6 | 13 | 13 | 13 | 5 |
| Reptiles | 6 | 6 | 8 | 8 | 8 | 8 | 8 | 8 | 8 | 6 |

Figure 1. Pottential numbers of amphibians, birds, mammals, and reptiles by stand age before and after treatment.

## CONCLUSIONS

Results from FOREX consultation runs suggest that this integrated expert system can be used to evaluate the impact on habitat suitability for amphibians, birds, reptiles, and mammals for northern red oak and oak-hickory forest types. The system also will provide economic and harvesting results for chosen scenarios. The manager or landowner can then evaluate his/her decisions in an integrated manner.

Results for this demonstration suggest that precommercial and commercial thinning treatments that remove 10 to 30% of the stand do not alter the site enough to alter habitat suitability for species that found the site suitable prior to the harvest treatment. Results suggest that final harvest operations impact the habitat suitability the most. Final harvest operations remove a significant amount of vertical and horizontal structure that is found suitable by a large number of amphibians, birds, reptiles, and mammals. Results also suggest that removing all down and dead woody material from the site following any treatment is detrimental to wildlife habitat suitability. Results suggest that a few species of amphibians, birds, reptiles, and mammals find any seral state suitable for habitat and, thus, are not impacted by any harvesting treatment. The most important structural needs for most of the snakes are rocky cover, forest litter, and down and dead woody material on the forest floor.

Although we only considered precommercial, commercial, and final harvest treatments for the northern red oak forest cover type, results suggest that FOREX can be used to evaluate the impacts of alternative harvesting treatments on the suitability of wildlife habitat by making additional consultation runs. FOREX could also be linked to FIA inventory data to estimate and evaluate wildlife habitat suitability.

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# RESIDUAL TREE DAMAGE DURING SELECTION CUTS USING TWO SKIDDING SYSTEMS IN THE MISSOURI OZARKS

Robert L. Ficklin[1], John P. Dwyer[2], Bruce E. Cutter[2], and Tom Draper[3]

Abstract: Today, there is an interest in using alternative silvicultural systems like selection and two-aged management, because the public finds these systems more acceptable than clearcutting. However, repeated entries into forest stands to remove timber increase the risk of residual stand damage. Harvest techniques are desirable that (1) reduce the risk of stand damage and (2) are economically efficient. Animal logging provides a feasible alternative to conventional rubber-tired skidder harvesting systems used in the Ozarks.

Residual stand damage following selection harvests was examined for both (1) a rubber-tired skidder and (2) a mule logger, to provide a case study comparison of logging damage between the two systems. We found that only 7% of the residual trees >5" dbh were damaged by the mule-logging system while 22% of the residual trees were damaged by the skidder system. Conventional skidding destroyed three times as many residual trees as mule logging. Residual stocking was similar across both sites.

## INTRODUCTION

Due primarily to public concerns about timber production on public forest lands, land management agencies are now implementing uneven-aged management plans on many federal and state forests. Questions still remain, however, about the long-term sustainability of uneven-aged management systems. Some tree species can not be easily regenerated under a residual forest canopy, which can cause a change in the species composition of the future stand. Furthermore, damage to residual trees during the selection harvest operations may decrease the quality of residual trees and increase stand mortality through insect and disease infestation. Before uneven-aged management regimes are embraced as a sustainable option for forest management, these questions must be addressed.

The implications of residual stand damage on public forest lands are important, due to the multiple-use designation on these lands. Clearcutting as a means of stand regeneration has become disfavored partly because of the lack of flexibility for other uses. Brunson and Shelby (1992) observed a relationship between the amount of overstory removal and public disapproval. For hiking, camping and scenic uses, stands with more residual trees were consistently given preference over clearcut areas. While uneven-aged management and partial cutting practices offer a compromise between timber production and other uses, excessive damage to residual trees during harvesting may compromise the aesthetic value of the residual stand (Schmid and Mata 1993). For this reason it is desirable to implement low impact harvest procedures when safety and productivity requirements permit. By reducing the negative impacts of harvesting on non-timber related uses, it is possible to increase the overall level of timber production which is socially optimal on public forest lands (Ficklin and others 1996).

In some areas cable/skyline systems and helicopter logging are viable options for low impact logging, but these systems are not widely available in the Ozark region. The use of animals such as mules and horses for skidding is both cost effective and available in the Ozarks, although skilled animal skidding loggers are becoming more difficult to find. There is also very little scientific data about the relative impacts of animal versus conventional

[1] Graduate Research Assistant at the University of Missouri-Columbia.
[2] Associate Professors of Forestry at the University of Missouri-Columbia.
[3] Assistant District Forester-Clearwater District, Missouri Department of Conservation.
*Authors 1, 2 & 3 can be reached at 1-31 Agriculture-UMC, Columbia, MO 65211, and author 4 can be reached at the Missouri Department of Conservation, Rt. 1 Box 1002, Piedmont, MO 63957.

rubber-tired skidding. In the 1930's a study of damage to loblolly pine (*Pinus taeda* L.) and hardwoods showed 12% of the residual pines were damaged, while over 35% of the hardwoods were damaged (MacKinney 1934). This study is significant because draft horses were the conventional means of skidding at the time. Other information about animal skidding is primarily anecdotal, with the assumption being that animal systems are lower impact than skidders.

Animal skidding is a lower productivity system compared to conventional rubber-tired skidders, so the needs of the forest landowner will affect the suitability of animal systems for a specific use. When high production is not needed or desired, the use of animal skidding may have long-term benefits from the avoidance of residual stem damage. There is considerable variability in the degree and extent of residual tree damage in the scientific literature. This fact can likely be attributed to differences in tree species, time of year, harvest techniques and logger attitudes. Schmid and Mata (1993) observed a range from approximately 20% to over 50% of residual lodgepole pine (*Pinus contorta* Dougl. Ex Loud.) trees damaged following partial cuts in Colorado and Wyoming. The authors suggest the thin bark of lodgepole pine is especially vulnerable to damage during the growing season; however, they further report that carelessness by skidder operators and poor sale administration attributed to the large number of damaged trees.

A study of residual damage to eastern white pine (*Pinus strobus* L.) in Ontario found that volume losses continued more than 10 years after the damage occurred (Whitney 1991). While 40% of the damaged trees were infected by decay-causing fungus, a full 90% of the damaged trees were stained. Although the stained wood is not structurally weakened like decayed wood, there is an economic loss associated with the discoloration. Projections of volume losses over 25-30 years conservatively estimated a loss of 25-30% of gross merchantable volume.

Residual damage to hardwoods in the Mississippi Delta was found to be more severe following a partial cut than in the pine studies. Meadows (1993) found over 50% of the residual green ash (*Fraxinus pennsylvanica* Marsh.) and 60% of the sugarberry (*Celtis laevigata* Willd.) trees were damaged. Meadows suggested that losses due to butt rot may be significant due to the characteristic spread of butt rot in both green ash and sugarberry. Furthermore, epicormic branching may increase due to reduced vigor from root damage and increased stem exposure, further affecting the marketability of timber from the future stand.

There is some evidence to suggest that timber production goals on areas under uneven-aged management may not be compromised by mechanized logging. Studies in West Virginia hardwood stands have shown that most of the damaged and destroyed trees are less than 5.0 inches in diameter (Lamson and others 1985), so the net loss of residual basal area was only 6%. The authors also suggest that trees that are damaged during one cutting cycle can be given priority for removal in subsequent cutting periods, which minimizes the occurrence of multiple damage to individual trees. However, financial losses associated with the reduced value of the damaged trees was not quantified, and the economic loss to other non-timber related uses was not examined. The issue of long-term sustainability of quality (undamaged) growing stock is also important, because of the large proportion of advanced regeneration damaged during the mechanized harvesting operation. Importantly, the study suggests that conventional mechanized systems are suitable for performing selection and partial cuts with minimal short-term loss of residual stand timber value.

Our study examines the use of animals (mules) for reducing both the extent and severity of residual tree damage compared to a rubber-tired skidder operation. Using animals for skidding may be a desirable option for landowners with primary management objectives besides timber production. The short-term objective of this study was to determine if animal skidding offers an alternative to rubber-tired skidders for reducing residual stand damage. Currently, there are very few loggers who use animals for skidding on a full-time basis; however, if provisions are made during the administration of timber sales, supply and demand dictates that a niche market will develop. Furthermore, the present use of animal systems is proof that such systems are economically viable, but the level of productivity with such systems is inherently lower than for conventional mechanized skidding systems. The long-term objectives of this project are: (1) to determine if relationships exist between tree damage during timber harvest and growth and productivity of the post-harvest forest, and (2) to determine if a relationship exists between the

species of trees damaged during timber harvest and the future developments of tree rot and disease. These objectives can be accomplished through follow-up measurements on the residual trees that have been tagged and cataloged.

## METHODS

### Study Areas

Two upland oak-pine harvest sites under uneven-aged management regimes were selected in Reynolds and Wayne counties of the Missouri Ozarks. The primary tree species on both sites were black oak (*Quercus velutina* Lam.), white oak (*Quercus alba* L.) and shortleaf pine (*Pinus echinata* Mill.). Both timber sales were marked and administered by foresters with the Missouri Department of Conservation, and harvesting on both sites occurred between January and August 1995 with interruptions due to weather. One rubber-tired skidder logger harvested in Reynolds county on a tract of the Clearwater State Forest, and one animal (mule) logger operated on a tract of land owned by a private hunting club in Wayne county. Both loggers were made aware that a research study was in progress, because of concerns about the rebar which marked our permanent plot corners. However, neither logger knew what the project was about or who else was involved. Since we were unable to replicate this study with multiple animal and rubber-tired skidder loggers, extensive preliminary information was collected on stand conditions. Pre- and post-harvest stand conditions were measured on both sites, so inferences could be made about the relative levels of stand damage for both the animal and the mechanized skidder.

### Pre-Harvest Data Collection

Six 4/5$^{th}$-acre (132 ft x 264 ft) plots were established systematically from a random starting point on both study sites, and a minimum buffer area of 66 ft. was maintained between all the plots. The total harvest area was 24 acres on the animal skidded site and 19 acres on the conventionally skidded site; consequently, the 4.8 acres of plots on both sites encompassed a full 20% and 25% of the harvest areas, respectively. A full inventory was made of all trees 5 inches in diameter and larger across all of the study plots, and site indices and slope measurements were also made for each plot. Site indices for black oak ranged from 63 to 70 on the animal skidded site, while indices ranged from 62 to 73 on the conventionally skidded site. Slope measurements were measured across the steepest part of the plots, and no plots fell entirely in a summit or footslope position. Accordingly, slopes on the animal/ alternative site ranged from 14-27%, and slopes on the rubber-tired skidder site were a similar 16-26%.

### Post-Harvest Data Collection

Following the harvest operations, each plot was fully inventoried again, and a total of 861 remaining trees were tagged. Each residual tree was measured and classified by species, by type of damage, and by severity of damage. The extent of stand damage was determined by the frequency of injury to residual trees, and the severity of damage is expressed as the area of cambium exposed. These measurements will be needed to monitor the difference in growth rates and the occurrence and severity of rot due to damage in accordance with the long-term objectives of this study. The residual trees that incurred no damage will be used for the estimation of changes in growth rate between damaged and undamaged trees.

There are 4 categories of damage: none, skidding, felling and destroyed. Skidding damage to roots or the bole was measured as the area of cambium exposed ($cm^2$). The stand damage from felling was measured in two ways depending upon the type and location of damage. Felling damage which exposed the cambium in the butt log (8 ft) was measured in the same manner as skidding damage. Crown damage resulting from felling was estimated as a fraction (quarters) of the crown destroyed. The degree of crown damage is not included in the cambium injury measurements, but crown losses will be considered in the long-term evaluation of changes in overstory productivity.

Volume estimates were calculated using a database management system developed by Seamon (1995) for the Missouri Department of Conservation. This system is designed to work with TWIGS (Miner and others 1988) and to convert data files for modeling. Volumes were calculated for sawtimber (bd ft) and for total cubic feet of volume,

including sawtimber, pulpwood and residue. The sawtimber values are the better estimate of the volume skidded through the stand, because most of the residues were not taken during skidding, but total volume does estimate the total volume of sawtimber, pulpwood and residue on the study sites.

Areas adjacent to access roads were not included in the sample plots on either site, due to the expectation of damage to trees along the roads. Because of the random location of sample plots within the harvest areas, the distance to the roads and log decks varied. To account for this variability, the area of both primary and secondary skid trails was measured within each plot. Access roads were closer to the plots on the site that was logged with rubber-tired skidders, so the skid trail lengths averaged 21% and 28% longer on the mule-skidded site for both primary and secondary trails, respectively. Width measurements were taken post-harvest in 10-ft intervals with a rolling tape, and total skid trail area was determined by a summation of the area of the resulting parallelograms. Because an intermittent drainage dissected one plot on the mechanized skidder study site, it was not possible to accurately measure the area of trails within that plot. This restriction was due to the felling of trees with the tops toward the drainage, which completely covered much of the area within the plot. As a result, the trail measurements on the mechanized skidder site are likely underestimated. Despite this limitation, the plot was included in the trail area calculations as having no skid trails, so that the conventional system would not be negatively biased in any way.

## RESULTS

The cut trees, volume and basal area per acre were determined by the difference between the pre-harvest plot observations and the post-harvest measurements. Table 1 provides a summary of the post-harvest plot conditions.

Table 1. Plot conditions post-harvest (expressed per acre)

| Animal skidder: | Plot 1 | Plot 2 | Plot 3 | Plot 4 | Plot 5 | Plot 6 | Mean/Median |
|---|---|---|---|---|---|---|---|
| Residual trees/ acre | 76 | 112 | 66 | 46 | 111 | 105 | 86/90.5 |
| Cut trees/ acre | 43 | 45 | 56 | 62 | 77 | 47 | 55/51.5 |
| Residual basal area (sq ft) | 59 | 63 | 44 | 37 | 59 | 64 | 54.3/59 |
| Cut basal area (sq ft) | 33 | 38 | 46 | 58 | 44 | 25 | 40.7/41 |
| Residual volume (bd ft) | 5766 | 4424 | 3851 | 4652 | 4189 | 6238 | 4853/4538 |
| Residual volume (cu ft) | 1800 | 1803 | 1297 | 1155 | 1657 | 1849 | 1594/1729 |
| Cut volume (bd ft) | 3637 | 3624 | 4397 | 6130 | 3235 | 1629 | 3775/3631 |
| Cut volume (cu ft) | 1054 | 1236 | 1410 | 1818 | 1274 | 699 | 1249/1255 |

| Rubber-tired skidder: | Plot 1 | Plot 2 | Plot 3 | Plot 4 | Plot 5 | Plot 6 | Mean/Median |
|---|---|---|---|---|---|---|---|
| Residual trees/ acre | 80 | 80 | 75 | 82 | 135 | 108 | 93.3/81 |
| Cut trees/ acre | 40 | 22 | 43 | 24 | 10 | 24 | 27.2/24 |
| Residual basal area (sq ft) | 67 | 65 | 52 | 62 | 58 | 59 | 60.5/60.5 |
| Cut basal area (sq ft) | 36 | 16 | 35 | 32 | 16 | 24 | 26.5/28 |
| Residual volume (bd ft) | 6183 | 6139 | 4186 | 5592 | 2607 | 4113 | 4803/4889 |
| Residual volume (cu ft) | 2134 | 2049 | 1539 | 1976 | 1600 | 1684 | 1830/1830 |
| Cut volume (bd ft) | 4100 | 1871 | 3485 | 4024 | 2058 | 2240 | 2963/2863 |
| Cut volume (cu ft) | 1170 | 533 | 1090 | 1109 | 443 | 861 | 868/976 |

Note: Cubic foot volumes include residues, and board foot volumes are for sawtimber material only.

The study plots on both sites encompassed group openings (~70 ft diameter) and individual-tree selection areas, as evidenced by the differences in basal area and volume across the plots. Neither the animal-skidded site nor the rubber-tired skidder site was consistently higher in all the plot parameters; however, the mechanized skidder site averaged 7 more trees and 6.2 $ft^2$ more basal area per acre than the mule skidded site. A Wilcoxon rank test was performed to compare the medians of plot measurements between the two study sites. The nonparametric rank test was used because of the small sample size (6 plots per site) and because of concerns about non-normal distribution. The only assumptions underlying the Wilcoxon test are independent observations, continuous random variables, at least ordinal measurement and equal variance between the independent populations (Daniel 1978). Siegel-Tukey tests were performed prior to the Wilcoxon tests to assure that there were no significant differences between population variances. Subsequent Wilcoxon test results indicate that residual trees per acre (p=.630), residual sawtimber volumes (p=.749) and residual basal area (p=.420) were not significantly different between sites. In absolute volumes, the measured difference in residual sawtimber volume was only 50 bd ft per acre higher on the animal-skidded plots than on the rubber-tired skidded plots. Based on these observations, we conclude that neither site was predisposed to greater damage based on residual stocking conditions.

Significantly, more trees were cut on the animal skidded tract than on the tract skidded conventionally with rubber-tired skidders (p=.005). There was slightly higher variability in basal area and volume cut among the plots, but the median basal area cut per acre on the animal-skidded tract was significantly higher (p=.037). While no difference in cut sawtimber volume (bd ft) was detected (p=.423), there was an additional 812 bd ft per acre removed from the animal-skidded site, and there was a higher volume (cu ft) of material felled on the animal-skidded site (p=.078). The significantly higher number of trees removed, the higher basal area removed and the higher cubic foot volume cut from the animal skidder site would be expected to increase the chance of damage to residual trees from both skidding and felling, since residual stand volumes were comparable.

The area of skid trails was an important contributing factor to residual stand damage. Both the primary and the secondary trails were longer on the mule-skidded site, due to the proximity of sample plots to the landings and access roads. However, the size and maneuverability of mechanized skidders resulted in wider skid trails compared to animals (Figure 1), so the overall area in both primary and secondary skid trails was significantly higher on the site logged with rubber-tired skidders when the area of each measured parallelogram was compared (p=.0001) for both trail types. The images in Figure 1 were taken at different times, so some interpretation is needed. The trails on the animal skidded site were photographed during the completion of the harvesting operation, whereas the trails on the mechanized skidder site were photographed 9 months after the harvest had been completed. As a result new regeneration can be seen on the mechanized skidder site but not on the animal skidded site. The difference in trail width, however, is readily visible.

The total area of skid trails measured within the 4.8 acres of plots on the animal-skidded site was 4,444.7 $ft^2$ for primary trails and 2,027.5 $ft^2$ for secondary trails—2.1% and 1% of the total harvest area, respectively. On the mechanized skidder site primary trails covered 10,704.5 $ft^2$ and secondary trails encompassed 9,627.5 $ft^2$—5.1% and 4.6% of the total area, respectively. This difference in trail areas was particularly significant, because the skid trail systems were longer on the site logged with animals. The primary and secondary trails extended 840 ft and 1,050 ft, respectively on the animal site, while the trails on the mechanized site were 670 ft and 760 ft long for primary and secondary trails, respectively. Despite the longer trail system and the inability to measure skid trails on one plot of the mechanized skidder site that we mentioned earlier, there was still less soil disturbance on the mule-skidded site. Since additional skidding area would be expected to cause greater damage to residual trees, the reduction of skid trail area is one of the main advantages of animal skidding. Similarly, residual damage would be expected to be minimized under mechanized skidder systems which use equipment that is not excessively large for accomplishing the removal of timber.

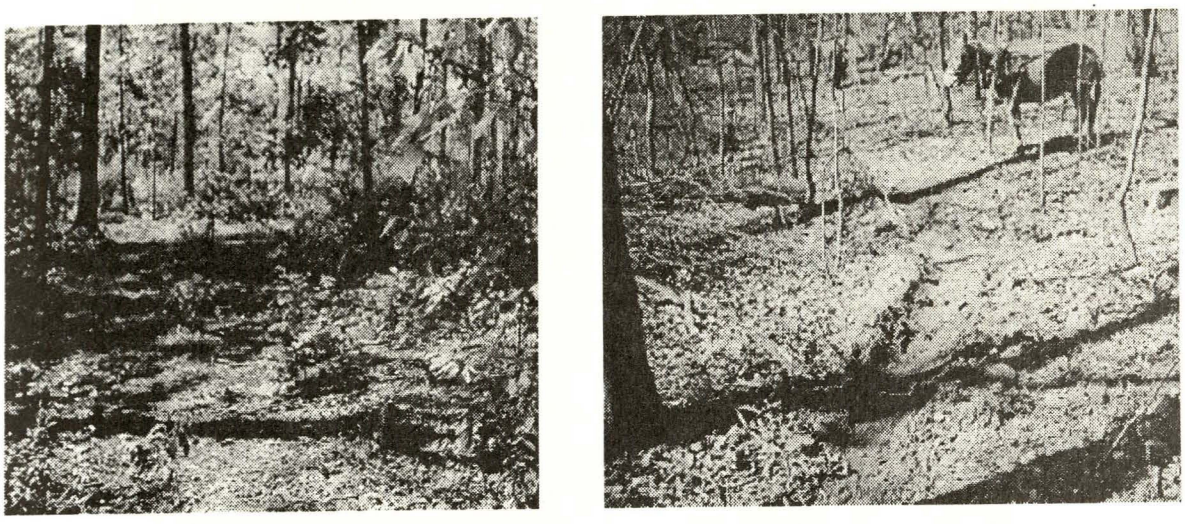

Figure 1. Primary skid trails-mechanized skidding [left] and animal skidding [right].

The species composition of the residual stands are summarized in Figure 2. Management objectives on the animal-skidded tract required an emphasis on shortleaf pine regeneration. Consequently, the residual percentage of pine on the alternative site was 26% compared to 1% on the conventional site. The largest component of both stands consisted of oaks [(white, black, and scarlet (*Quercus coccinea* Muenchh.)], and the proportion of other species, including hickories (*Carya* spp.), sugar maple (*Acer saccharum* Marsh.) and dogwood (*Cornus florida* L.), was similar between the two study locations.

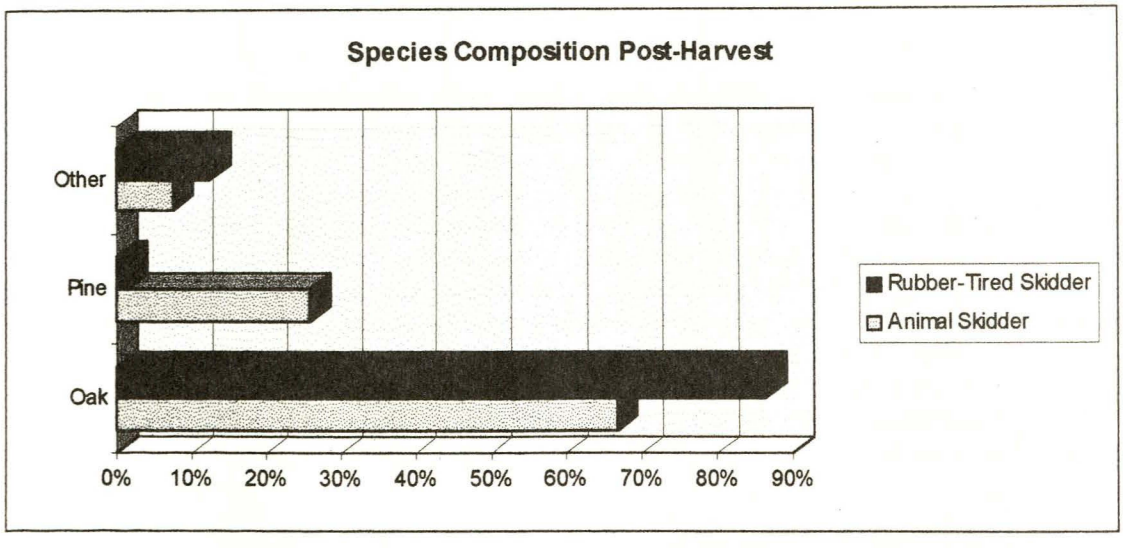


Figure 2. Comparison of species composition.

Damage was most severe on the conventionally skidded plots (Table 2). A Wilcoxon test detected a significantly higher median percentage of damaged trees (p=.025) and damaged cambium area (p=.055) per acre on the rubber-tired skidder site. The area of cambium damage is higher on the rubber-tired skidder site, but when the total area of cambium exposed is divided by the number of total residual trees, there is an average damage per stem of 22 $cm^2$ and 36 $cm^2$ on the animal and conventional sites, respectively. When the area of damage is projected across only the damaged residual trees, there is 240 $cm^2$ of cambium exposed on 8 damaged trees per acre on the animal-skidded site. On the rubber-tired skidder site there is an average of 130 $cm^2$ of injury on 26 trees per acre. Three times as many trees were destroyed by either skidding or felling under the rubber-tired logging system as under the mule-skidder system.

Table 2. Summary of residual tree damage

| Animal skidder | | |
|---|---|---|
| | Residual trees damaged (%) | Area of cambium damage (sq cm) |
| Plot 1 | 12 | 1,848 |
| Plot 2 | 9 | 7,194 |
| Plot 3 | 9 | 1,118 |
| Plot 4 | 11 | 305 |
| Plot 5 | 5 | 998 |
| Plot 6 | 2 | 50 |
| Mean/Median | 8/9 | 1,919/1,058 |

| Rubber-tired skidder | | |
|---|---|---|
| | Residual trees damaged (%) | Area of cambium damage (sq cm) |
| Plot 1 | 33 | 4,384 |
| Plot 2 | 19 | 1,628 |
| Plot 3 | 28 | 8,429 |
| Plot 4 | 23 | 2,231 |
| Plot 5 | 8 | 1,957 |
| Plot 6 | 30 | 1,992 |
| Mean/Median | 23.5/25.5 | 3,439/2,112 |

The breakdown of wounds by damage type is given in Figure 3. The percentages represent the fraction of damaged trees in each category, so similar percentages do not indicate similar absolute numbers. The greatest source of stand damage on both harvest areas came from skidding, which accounted for 43% of animal skidding and 47% of conventional skidding damage. Felling damage was also a key source of damage on both areas, and the proportion of trees damaged by felling was similar between the sites; however, a greater actual number of trees was damaged by felling on the rubber-tired skidder site. Felling prior to the animal skidding operation caused 40% of the total stand damage. Similarly, 38% of the damage on the rubber-tired skidder tract was due to felling. The summary of tree damage provided in Table 3 gives the absolute number of trees damaged across the study plots on a per-acre basis.

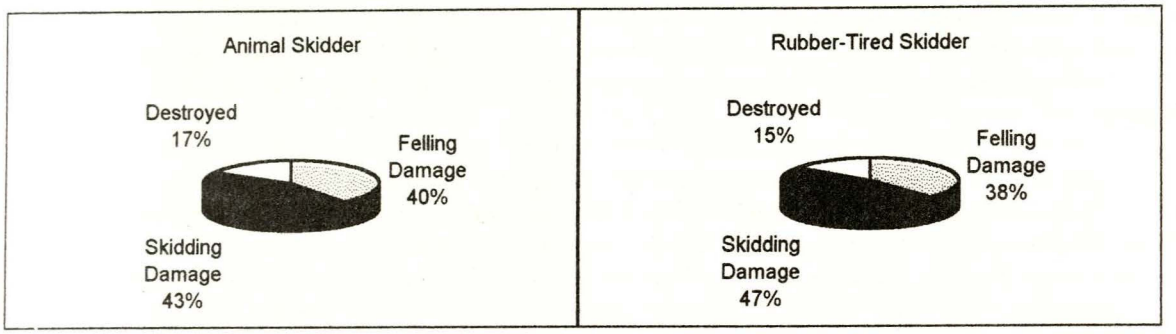


Figure 3. Residual tree damage by damage category (% of damaged trees).

The proportion of damaged trees that were destroyed was also similar between the tracts with 17% and 15% of the damaged trees destroyed on the animal and mechanized sites, respectively. Although the proportions of damaged trees in each category are similar, the actual number of damaged trees was higher in all damage classes using a rubber-tired skidder. Therefore, some component of the mechanized system other than skidding alone contributed to residual damage. One possible explanation would be the higher rate of production under the mechanized system. Although it was not possible to perform an analysis of operating costs between these two systems, it is certain that the overhead costs of the mechanized system were higher than for the animal-skidder system. The animal logger only had a truck and chainsaw, and the logger worked alone. Conversely, the mechanized system had an additional cost for skidder equipment, which would necessitate higher production to meet operating costs.

Table 3. Breakdown of tree damage per acre by category

| | Animal skidder | Rubber-tired skidder |
|---|---|---|
| Felling damage | 3 | 8 |
| Skidding damage | 3 | 10 |
| Destroyed | 1 | 3 |
| Total per acre: | 7 | 21 |

## DISCUSSION

In contrast to some past studies of residual stand damage, less than 25% of the residual trees were damaged by rubber-tired skidders, which suggests that the logger was both competent and conscientious. The contribution of a well administered sale cannot be discounted, either. Regardless, the results found on the animal skidded site were a sharp contrast to the rubber-tired skidder area. Skidding and felling damage were both much less frequent on the animal skidded site, and there were three times as many trees destroyed with the mechanized skidding system. Stand stocking conditions post-harvest were not significantly different between the sites, but the pine component was different between the sites.

Because more trees were damaged by skidding on the conventionally [mechanized] skidded tract, more surface area of cambium was exposed (Table 2), and a greater loss of merchantable volume will be expected. Future re-measurements of tagged trees, damaged and undamaged, can permit the estimation of losses due to decreased growth rate and disease. Although a greater surface area was exposed by conventional skidding, the actual severity of tree

damage by rubber-tired skidders was not greater per residual tree. The average area of injury per stem is actually lower on the conventionally logged site (130 $cm^2$ vs. 240 $cm^2$), which suggests that rubber-tired skidders caused injuries no more severe than the mules caused. Despite the smaller surface area, it would be inappropriate to conclude that the rubber-tired skidder did less damage, because so many fewer trees were damaged on the animal skidded site.

One similarity between the systems was the proportion of damage in each damage category. This fact cannot be explained by the difference between an animal and a rubber-tired skidder, since the felling damage and most of the destroyed trees were injured prior to skidding. Two key factors can be identified as the cause for the increased felling damage on the mechanically logged site. First, the higher productivity of the rubber-tired skidder system does not allow previously felled material to be removed prior to the completion of felling. On the mule-skidded site, the logger removes fewer logs per day; consequently, the mule logger gradually opens up the stand by cutting fewer trees at one time and removing the material. On the rubber-tired skidder site more trees have to be felled at one time to supply the skidder, putting the residual trees at higher risk of felling damage. This hypothesis was not tested, but it does offer a possible explanation for the increased damage in all categories.

The species composition of the residual stands is the second factor contributing to the increase in felling damage. The morphological characteristics of shortleaf pine and oaks affect their susceptibility to both skidding and felling damage. As Figure 2 shows, there was a significant pine component on the animal skidded site, but only 1% of the residual trees on the rubber-tired skidded site were pine. Because of the self-pruning of pine, there were fewer lower branches to be damaged by falling trees. There was only one occurrence of felling damage to pine across both sites (Figure 4) which suggests that pine were less susceptible to damage during felling. These results are similar to those observed by MacKinney (1934) in stands of loblolly pine and hardwoods following harvesting with draft horses.

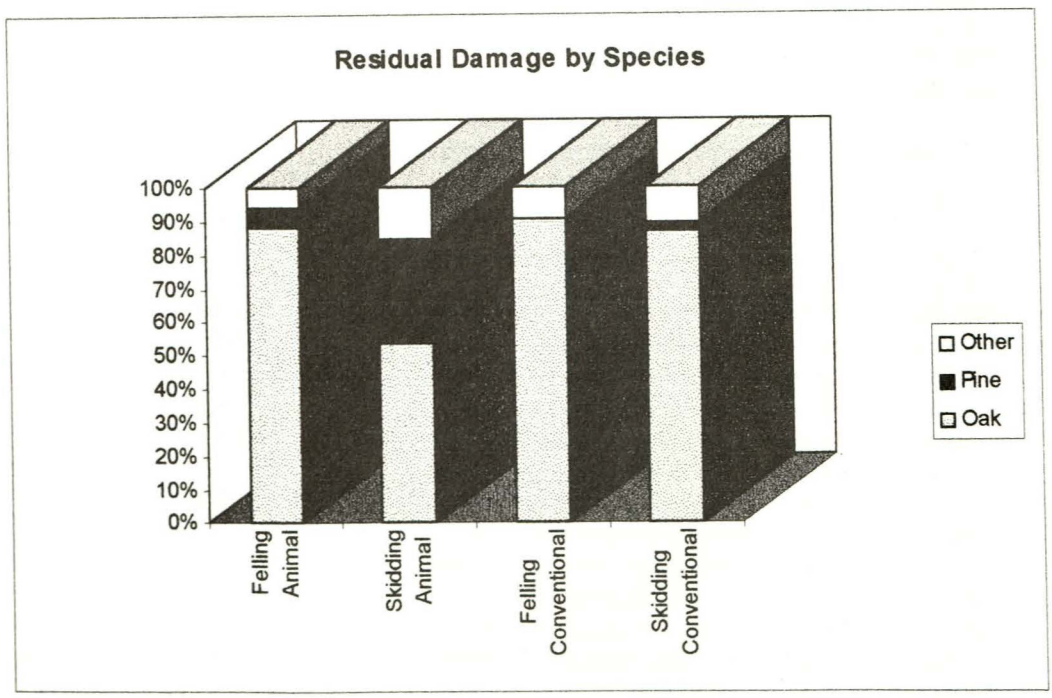


Figure 4. Residual tree damage by species.

Lateral branches on the oak and other non-pine species were an obstacle during felling, so the higher incidence of felling damage on the conventional site is partly explained by the lack of a significant pine component. However, more than three times as many oak and other hardwoods were damaged during felling on the mechanized study area compared to the animal skidded area, so some aspect of the mechanized skidding system must account for the discrepancy in felling damage between the sites. Under the higher production of the rubber-tired skidder operation, our results indicated that more trees were likely to be damaged during felling, so careful planning of roads and skid trails should be done to help minimize felling damage. Skidding damage, unlike felling damage, did not appear to be dependent upon species in this study, because the proportion of pine and other species damaged by skidding was similar to the composition of the residual stand.

## CONCLUSIONS

Although this case study cannot conclusively state that animal skidding reduces residual stand damage under all conditions, the summary statistics do suggest that the mule skidding operation resulted in less damage to the residual stand. The reduced damage on the alternative harvest area may be partly attributable to logger care and stand composition, but the animal skidding system was the most significant reason for the reduced damage. Residual plot measurements of trees per acre, volumes and basal areas were similar between the two study areas, but the animal skidder removed more trees and more of the standing volume. With similar residual stocking densities between sites, removing more trees from one stand would be expected to increase the level of stand damage. However, there was a much lower frequency and severity of damage on the mule skidded site which had the most timber removed.

The area disturbed by skidding was also a factor that had a sharp contrast between the two skidding systems. Only 3.1% of the animal skidded tract was disturbed during harvesting, but a minimum of 9.7% of the area logged with rubber-tired skidders was disturbed. This relationship implies that residual damage may be reduced by minimizing the size of machinery used during conventional logging operations. Given the limited availability of loggers who use animals, it is important to apply the results of this and other studies to the standard harvest methods that are available. However, when animal skidding operators are available, there is a clear advantage of using animals to reduce the area disturbed during harvesting.

For forest landowners with primary management objectives other than timber production, animal logging may be a viable option for minimizing residual stand damage. If animal skidding is available, our results suggest that this alternative logging system is effective in reducing tree damage and soil disturbance. Timber cutting can be an important component of any wildlife or recreation management plan, and alternative logging systems could mitigate concerns over the impact of harvesting. Furthermore, the use of effective low-impact logging systems in a multiple-use management paradigm may increase the level of timber production on public lands which is socially optimal. The long-term effects of residual stand damage on site productivity is still unclear, but minimizing stand damage through effective sale administration and appropriate harvest techniques is an achievable and worthy goal.

## ACKNOWLEDGEMENTS

The authors extend their appreciation to the Mountain Lake Hunting & Fishing League for permitting the study of harvest operations on their land. We would also like to thank the University of Missouri System Research Board and the Missouri Department of Conservation for funding this project. Furthermore, we would like to thank the MDC Clearwater Forest District foresters for administering the sales and the peer reviewers for their helpful comments.

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# PERCEIVED IMPACTS OF DISTURBANCE IN CENTRAL HARDWOOD ECOSYSTEMS

John Hetherington and John Burde[1]

Abstract: The intent of this research was to empirically identify and estimate the relationship between perceived scenic beauty and biophysical resources of hardwood stands in southeast Missouri under various disturbance regimes. The scenic effects of different harvest methods, wind, and fire were systematically investigated using a psychophysical approach. Ratings of scenic beauty were collected from both residents and non-residents, and scaled using the Scenic Beauty Estimation (SBE) method. Quantitative, predictive models were generated to identify what biophysical features contributed to public perceptions of scenic beauty. In addition, information concerning familiarity and experience with the area, group membership, knowledge about natural resources, and demographics were collected to assess how the perceptions of scenic beauty of disturbances related to respondent characteristics.

[1] Department of Psychology, Southern Illinois University, Carbondale, IL 62901-6502.

# NITROGEN BUDGETS ON APPALACHIAN FOREST CATCHMENTS

David R. DeWalle[1]

Abstract: Variations in nitrogen losses in streamflow on catchments in the Appalachians suggests that the level of nitrogen retention in hardwood forests varies widely. Stream losses of dissolved nitrate-N on several small experimental forested catchments range from about 0.2 to 8.5 kg ha-1 y-1. This wide range of losses is equivalent to less than 10% to nearly 100% of measured atmospheric nitrogen wet deposition in the region. Forest stand age and condition and levels of atmospheric wet N deposition do not correlate well with observed differences in catchment N losses. Variations in soil mineralization and nitrification rates and soil C/N ratios appear to be one cause of variations. Ammonium-N export from forested catchments is quite low and can generally be ignored. Organic-N export is not often evaluated, but estimates indicate that export of dissolved organic-N in streamflow may be as important as nitrate-N losses. Organic-N losses may be quite sensitive to forest management activities and should be evaluated for accurate nitrogen budgets. Improved predictions of nitrogen losses on forest land are needed for models of water quality in receiving waters such as the Chesapeake Bay and for prediction of forest land response to changing levels of atmospheric deposition.

[1] School of Forest Resources and Environmental Resources Research Institute, The Pennsylvania State University, University Park, PA 16802.

# SPATIAL PATTERNS OF *ARMILLARIA* POPULATIONS IN THE WALKER BRANCH WATERSHED THROUGHFALL DISPLACEMENT EXPERIMENT, TENNESSEE, USA.

Johann N. Bruhn[1], James A. Brenneman[2], James J. Wetteroff, Jr.[1], Jeanne D. Mihail[1], and Theodor D. Leininger[3]

Abstract: Species in the white-rot fungal genus *Armillaria* vary in parasitic aggressiveness as root and butt rot pathogens of trees. *Armillaria* genets (individuals) were mapped in the Throughfall Displacement Experiment (TDE) using mushrooms and rhizomorphs collected in 1994 and 1995. Initiated in July 1993, the TDE consists of three 80 x 80 m plots. The "ambient" plot receives natural precipitation; the "wet" plot also receives 33% of throughfall from the "dry" plot (which also had lower pre-treatment mean soil water content). Isolates from 36 rhizomorph and 48 mushroom collections of *A. gallica* Marxmüller & Romagnesi, and from 21 mushroom collections of *A. mellea* (Vahl:Fr.) Kummer, represent 5 *A. gallica* genets and 18 *A. mellea* genets. Only 3 of the 84 *A. gallica* collections, and 1 of the 21 *A. mellea* collections, were obtained from the dry plot. The two largest *A. gallica* genets occupy most of the wet and ambient plots, overlapping very little. Most of the *A. mellea* genets are represented by single collections throughout the wet and ambient plots. At least nine *A. mellea* genets occur within the boundaries of the two largest *A. gallica* genets. *Armillaria* species interactions with each other, with forest community vegetation, and with the TDE treatments are hypothesized.

## INTRODUCTION

As atmospheric levels of greenhouse gases increase, climate modelers predict a rise in average global temperature and associated changes in precipitation patterns. Although greater climatic instability, including more frequent summer droughts, has been predicted (Kraüchi 1993, Rind and others 1990), there is some uncertainty whether growing season precipitation within the central hardwood forest region will increase or decrease. To test the responses of a central hardwood forest to both reduced and increased growing season precipitation, the Throughfall Displacement Experiment (TDE) was established on the Walker Branch Watershed, near Oak Ridge, Tennessee (Hanson and others 1995). Pre-treatment soil moisture observations were initiated in April 1992, and treatments were imposed in mid-summer 1993. Treatments consist of transferring one-third of throughfall precipitation from the "dry" plot, across the "ambient" plot, to the "wet" plot. It was anticipated that the dry treatment would reduce soil moisture to approximately the same extent as the driest season of the 1980's drought (Cook and others 1988). Studies of *Armillaria* populations on the TDE plots were initiated in 1994 to evaluate TDE treatment effects on the behavior of this ubiquitous genus of root parasitic wood decay fungi. *Armillaria* spp. (Tricholomataceae, Agaricales) often serve as pivotal contributing factors in stress-mediated forest declines (Manion and Lachance 1992, Wargo 1984). Both excess moisture and drought during the growing season have been implicated as factors inciting Armillaria root disease (e.g., Lonsdale and Gibbs 1996, Wargo and Harrington 1991).

*Armillaria* (Fr.:Fr.) Staude is a genus of agaric fungi which obtain energy primarily through the white-rot mode of wood decay, metabolizing both cellulose and lignin in the process (Garraway and others 1991). Wood decay is accomplished by the exploitive activity of a highly branched network (mycelium) of microscopic filamentous tubes (hyphae). A portion of the energy derived from wood decay is spent on sexual reproduction. *Armillaria* spp. reproduce by the production of airborne spores on the gills of basidiomata (mushrooms). Each spore which successfully germinates and colonizes a suitable woody substrate (foodbase) generally mates with another sexually compatible germling to form a genetically unique individual (genet), which may establish itself in the landscape as an agent of wood decomposition. *Armillaria* spp. do not produce asexual spores. Instead, genets spread through the forest

[1]Department of Plant Pathology, 108 Waters Hall, University of Missouri, MO 65211.
[2]Department of Biology, University of Evansville, Evansville, IN 47722.
[3]Southern Research Station, Southern Hardwoods Laboratory, P.O. Box 227, Stoneville, MS 38776.

floor by the foraging growth of exploratory organs called rhizomorphs, which are branching networks of cord-like hyphal aggregates. Because most of the resources required to fuel rhizomorph growth are provided by decay of the source foodbase, the aggressiveness of *Armillaria* activity is directly related to the distribution and abundance of woody foodbases fueling rhizomorph growth (Garrett 1956). When a rhizomorph encounters an appropriate foodbase, it penetrates the bark and differentiates back into a mycelium. Thus, a single genet has the potential to occupy numerous foodbases simultaneously, and to maintain physical continuity indefinitely within the area of forest floor it occupies. Dead or severely stressed roots of living trees may be colonized in this manner, and live roots may have their resistance tested repeatedly by rhizomorphs. Established genets appear to have a preemptive advantage over invading conspecific genets in acquiring foodbase resources which become available within their boundaries. As a result, it appears that survival of new genets is very poor within the bounds of established genets of the same species (Smith and others 1992, 1994). Therefore, genet maps of a single *Armillaria* sp. often form a mosaic of non-overlapping pieces of various sizes. Perhaps due to differences in host specialization and virulence, genet mosaics of different *Armillaria* spp. often occupy the same area (e.g., Anderson and Kohn 1995, Kile and others 1991, Rizzo and Harrington 1993). *Armillaria* spp. vary in their relative aggressiveness toward different kinds of trees, and different genets of the same *Armillaria* sp. often differ in aggressiveness toward their preferred host species (Omdal and others 1995). A loose relationship has been noted between the rhizomorph growth characteristics of different *Armillaria* spp. and their relative ability to cause root disease (Morrison 1989). Thus, *A. gallica*, a prolific producer of rhizomorphs capable of establishing very large genets, is only very weakly pathogenic, whereas *A. mellea*, producing smaller rhizomorph systems and genets, is much more pathogenic (e.g., Gregory and others 1991, Rishbeth 1991).

Our immediate study objectives were to identify to species all *Armillaria* genets occupying the TDE plots, and to determine their spatial distributions with respect to each other and the TDE design. Long-term objectives are: to (1) determine the influences of TDE treatments on wood decay, rhizomorph initiation and growth, and fruiting by *A. gallica* and *A. mellea*, (2) determine the ecology of competition between genets of *A. gallica* and *A. mellea* as affected by the TDE treatments, and (3) evaluate the epidemiology of Armillaria root disease on the TDE plots.

## METHODS

### Site Description

The Throughfall Displacement Experiment (TDE) is located just below the top of a south-facing slope (approximately 30%) of the Walker Branch Watershed (35°58' N latitude, 84°17' W longitude) near Oak Ridge, Tennessee. Mean annual precipitation and temperature are 140 cm and 13.3° C, respectively (Hanson and others 1995). Soils at the TDE site are primarily typic Paleudults: acidic (pH 3.5-4.6), cherty, infertile, permeable, approximately 30 m deep overlying dolomitic bedrock (Hanson and others 1995). Tree cover is dominated by *Quercus prinus* L., *Q. alba* L., *Nyssa sylvatica* Marsh. and *Acer rubrum* L. (Table 1, raw data provided by P. J. Hanson, Oak Ridge National Laboratory, Environmental Sciences Division, Oak Ridge, TN).

### Study Design

The TDE. The TDE site is divided into three contiguous 80 x 80 m treatment plots (designated dry, ambient, and wet, from east to west) along the upper slope of the watershed. Each plot is subdivided into 100 subplots for sampling purposes. Precipitation throughfall reaching the forest floor is manipulated by a system of suspended subcanopy troughs in the dry plot which intercept approximately 33% of canopy throughfall, coupled to a system of pipes which transfer the intercepted throughfall by gravity flow across the ambient plot and distribute it uniformly onto the wet plot (Hanson and others 1995). Soil moisture in the upper 35 cm is monitored by time domain reflectometry (TDR) at 310 sample locations across the TDE site. Soil matric potentials are estimated from the TDR data using moisture retention curves generated for the TDE site soils. Pre-treatment evaluations of seasonal soil water content indicated significant spatial patterns: soil water content decreased with increasing elevation and was lower in the eastern one-third of the site than in the remaining two-thirds. Because these spatial patterns were seasonally consistent, they have been captured in a covariance matrix which permits analysis of the effectiveness with which the TDE influences soil water contents of the wet and dry treatment plots with respect to the ambient plot (Hanson and others 1995).

Table 1. Characteristics of the tree cover (stems >10 cm dbh) on the Walker Branch Watershed Throughfall Displacement Experiment site as of May 1991[1].

| | Basal Area ($m^2 \cdot ha^{-1}$) | | | Stems ($n \cdot ha^{-1}$) | | |
|---|---|---|---|---|---|---|
| Species[2] | Wet | Ambient | Dry | Wet | Ambient | Dry |
| *Quercus prinus* L. | 5.3 | 3.3 | 8.3 | 56.2 | 25.0 | 68.8 |
| *Quercus alba* L. | 6.4 | 4.7 | 2.0 | 53.1 | 64.0 | 21.9 |
| *Nyssa sylvatica* Marsh. | 1.9 | 3.8 | 2.9 | 51.6 | 118.8 | 95.3 |
| *Acer rubrum* L. | 2.4 | 2.6 | 2.5 | 68.8 | 53.1 | 65.6 |
| other *Quercus* spp. | 2.2 | 1.5 | 1.7 | 20.3 | 14.1 | 12.5 |
| *Liriodendron tulipifera* L. | 1.3 | 1.4 | 2.3 | 14.1 | 17.2 | 31.2 |
| *Oxydendrum arboreum* (L.) D.C. | 0.5 | 1.0 | 1.0 | 20.3 | 50.0 | 29.6 |
| *Acer saccharum* Marsh. | 0.4 | 1.0 | 0.3 | 7.8 | 9.3 | 6.3 |
| *Carya* spp. | 0.6 | 0.3 | 0.4 | 7.8 | 3.1 | 3.1 |
| *Cornus florida* L. | 0.0 | 0.3 | 0.3 | 0.0 | 18.8 | 20.3 |
| *Prunus serotina* Ehrh. | 0.1 | 0.1 | 0.1 | 3.1 | 6.3 | 6.3 |
| conifers ( *Pinus, Juniperus*) | 0.0 | 0.3 | 0.1 | 0.0 | 3.1 | 7.8 |

[1]Raw data provided by P.J. Hanson, Oak Ridge National Laboratory, Environmental Sciences Division, Oak Ridge, TN.
[2]Trees >10 cm dbh on the experimental site represent 18 species; nine of these species were grouped into three categories for representation in this table.

*Armillaria* Population Studies. The forest floor surface in each TDE subplot was repeatedly searched during the *Armillaria* fruiting seasons (mid-September through early-November) of both 1994 and 1995 for *Armillaria* basidiomata and for rhizomorphs colonizing woody debris. All collections were mapped. *Armillaria* isolates were identified to genet based on vegetative compatibility tests among the collected field isolates (Adams 1974, Guillaumin and others 1996, Smith and others 1994). Vegetative compatibility was determined through confrontational pairings of isolates on agar-solidified (2% w/v) malt extract (3% w/v) growth medium. Pairings were duplicated for confirmation of results. Isolates belonging to the same genet are vegetatively compatible and grow together freely, whereas isolates representing different genets (regardless of species) are vegetatively incompatible and form a visible reaction at close proximity. A diploid field isolate representing each *Armillaria* genet was then used to identify that genet to species in mating tests with voucher single-basidiospore (haploid) isolates of *Armillaria* spp. (Korhonen 1978, Rizzo and Harrington 1992 ). The voucher isolates used in these matings were verified and supplied to the senior author by J. B. Anderson (Department of Botany, Erindale College, University of Toronto).

A map showing approximate locations of *Armillaria* genets for the TDE site was produced, in which each genet is represented as a polygon containing all collection points for that genet and bounded by straight line segments which connect collection points. We have represented each genet as a spatially continuous individual, recognizing that individual genets may repeatedly fragment and rejoin depending on conditions of forest health perceived by the fungi as periods of famine or feast (Anderson and Kohn 1995; Garrett 1956; Rayner 1991; Smith and others 1992, 1994).

Statistical Analysis

For each *Armillaria* species, we tested the null hypothesis that the numbers of field isolates collected in the three TDE treatment plots were not significantly different from expected values ($\alpha = .05$). A chi-square test of a 3 X 1 contingency table (Sokal and Rohlf 1995) was used to test this hypothesis for each species. For each species, the expected number of isolates collected from each plot was one-third of the total number of isolates collected from all three plots.

## RESULTS

Covariate analyses of soil moisture data indicate that the TDE accomplishes both significant drying of the "dry" plot and significant wetting of the "wet" plot during both relatively dry and wet growing seasons (Hanson and others 1995). All *Armillaria* isolate collections were made from basidiomata and rhizomorphs associated with woody debris on the forest floor. Isolates from 36 rhizomorph and 48 basidioma collections of *A. gallica*, and from 21 basidioma collections of *A. mellea*, represent 5 *A. gallica* genets and 18 *A. mellea* genets. Basidiomata of *A. gallica* were collected between 13 October and 2 November 1994; basidiomata of both species were collected on 11 October 1995. While rhizomorph collection is not seasonally limited, most rhizomorph collections were also made in the late autumn.

Three points stand out from inspection of the *Armillaria* field isolate map (Figure 1). First, genets of *A. gallica* achieve much greater size than do genets of *A. mellea*, probably due to *A. gallica*'s greater production of rhizomorphs (see Redfern and Filip 1991). In this regard, we note that all rhizomorph collections from the TDE plots belonged to *A. gallica* genets. The two largest *A. gallica* genets were represented by isolates collected from 9 and 46 subplots, respectively, and appear to occupy approximately 0.33 ha and 0.54 ha, respectively. Fifteen of the 18 *A. mellea* genets were recovered from only one subplot (0.0064 ha) each; only three *A. mellea* genets were recovered from two adjacent TDE subplots (Figures 1 and 2).

Second, genets of *A. gallica* appear to represent continuous and territorial individuals, with little tendency to overlap one another. The large size of *A. gallica* genets "a" and "b" may explain the smaller number of *A. gallica* genets detected compared with *A. mellea*. While genets of *A. mellea* in the TDE plots are apparently too small to evaluate their territoriality, 11 of the 18 *A. mellea* genets appear to occur within the bounds of the two largest *A. gallica* genets.

Third, surprisingly few collections of either *Armillaria* species were made in the dry plot, compared with either the wet or ambient plots. The two largest *A. gallica* genets occupy most of the wet and ambient plots. Only one genet of each species, represented by 3 of the 84 *A. gallica* collections and 1 of the 21 *A. mellea* collections, was detected in the dry plot. For both *A. mellea* and *A. gallica*, the distributions of isolates among the three TDE plots differed significantly from expectation (Table 2). For both species, more collections than expected were made in the ambient plot, and fewer collections than expected were made in the dry plot.

## DISCUSSION

It is most appropriate to think of *Armillaria* populations as consisting of typically territorial, long-lived, stable genets of various sizes. This requires a conceptual adjustment for many forest scientists, since (superficially) genets are only apparent as scattered, seasonal, ephemeral, sporadically produced clusters of mid-size basidiomata (Watling 1995).

The spatial distributions and ecological attributes of *Armillaria* spp. genets in a forest community have important implications for forest community responses to perturbations ranging from silvicultural operations to climate change. Each *Armillaria* genet exercises ecological influence through its unique genetic potential for woody resource foraging, wood decomposition and nutrient recycling, and root parasitism, as mediated by environmental constraints on the expression of these potentials. Without supplemental controlled experimentation, it is difficult to establish (e.g., Bruhn and others 1994, 1996) whether the differences observed in behavior of neighboring

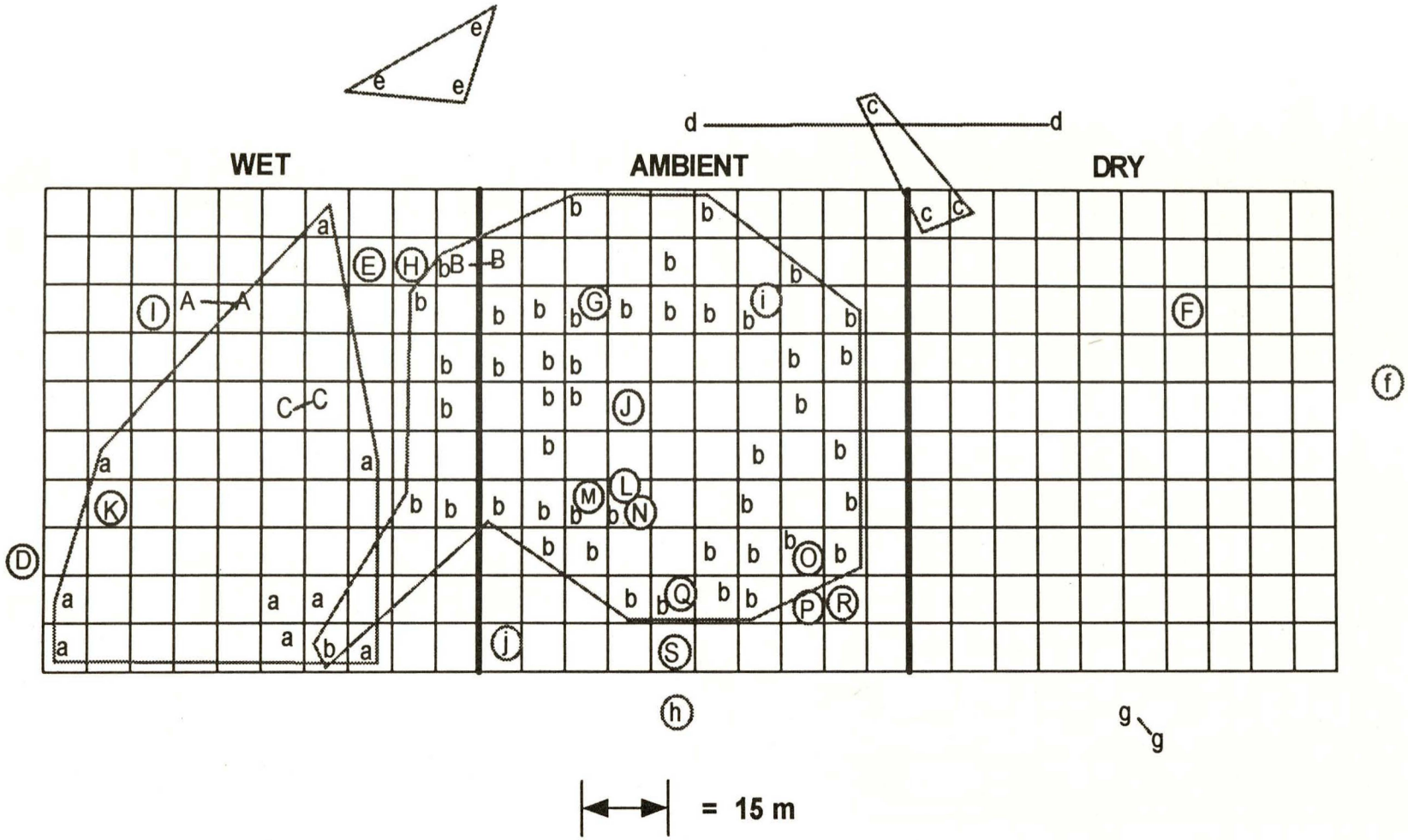


Figure. 1. Spatial distribution of *Armillaria* genets in the TDE plots. Each collection point is marked with a letter; lower-case letters indicate *A. gallica* isolates, upper-case letters indicate *A. mellea* isolates. Isolates belonging to the same genet share the same upper- or lower-case letter. Approximate boundaries of each genet are indicated.

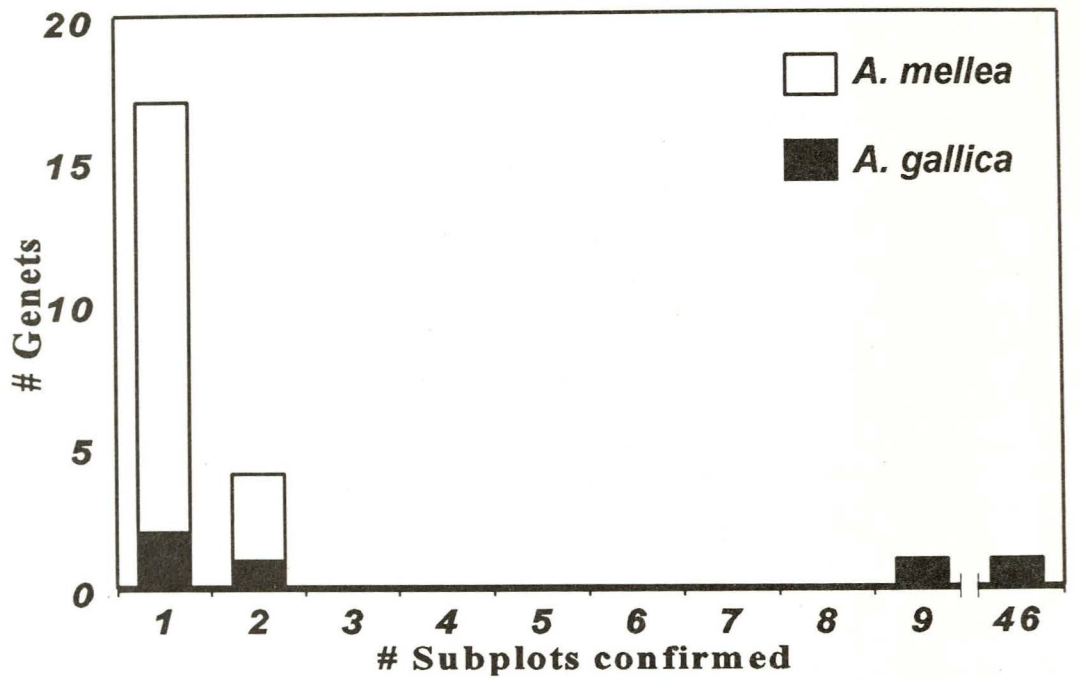


Figure 2. Relative sizes of *A. gallica* and *A. mellea* genets occupying the TDE plots, expressed as the number of genets of each species recovered from different numbers of subplots.

Table 2. Numbers of *Armillaria gallica* and *A. mellea* field isolates collected from the three TDE plots (for 1994 and 1995 combined), and the associated Pearson statistic ($X^2$ for goodness of fit) values. The tabular value of $\chi^2_{.05[2]}$ is 5.991.

| | | TDE plot | | | | |
|---|---|---|---|---|---|---|
| | | Wet | Ambient | Dry | Total | $\chi^2$ |
| *A. gallica* | Observed | 16 | 65 | 3 | 84 | 76.4 |
| *A. mellea* | Observed | 9 | 11 | 1 | 21 | 8.0 |

*Armillaria* genets are due to differences among those genets' genetic potentials (e.g., Omdal and others 1995) or to limitations of the local environment in which each genet functions (such as soil structure, distribution of suitable foodbase resources, interactions with competitors, etc.). Because *Armillaria* genets are fundamentally territorial, continuously distributed in the forest floor, and long-lived, they exercise their ecological influences in discrete portions of the forest landscape over time frames of centuries or even millennia (e.g., Anderson and others 1979, Shaw and Roth 1976, Smith and others 1992). In this way, the spatial pattern of *Armillaria* genets represents one component of a forest landscape's genetic "memory" of how to respond to change, thus helping to determine long-term forest community structure. As a result, the spatial pattern of *Armillaria* genets even influences wildlife populations through effects on vegetation structure.

Until more work is done, we can only speculate on the interactions occurring in the forest floor between *Armillaria* genets of the same or different species. Territorial behavior of con-specific *Armillaria* genets coupled with the apparent overlap of genets of different species has been noted previously (e.g., Kile and others 1991; Rizzo and Harrington 1993; Smith and others 1992, 1994). We assume that the unique genetic potential of each *Armillaria* genet is expressed exclusively within that genet's boundary, as appeared to be the case for *A. ostoyae* (Romagnesi) Herink genets in red pine plantations (Bruhn and others 1994, 1996). When genets of certain *Armillaria* spp. occupy the same landscape, it may be possible as a result of niche overlap (see Leibold 1995) for genets of the different species to influence each other's behavior in areas of genet contact or overlap. For example, since both *A. gallica* and *A. mellea* are hardwood-specializing fungi, and *A. gallica* is a much more aggressive producer of rhizomorphs, *A. mellea* may often have less access to the portion of foodbase resources suitable for both species within the boundaries of *A. gallica* genets, resulting in a smaller "realized niche" than elsewhere in the absence of competition from *A. gallica*. Thus, as a result of niche overlap, *A. gallica* might be expected to mitigate the expression of plant root parasitic virulence by a co-occurring *A. mellea* genet. On the other hand, because of their greater virulence as root parasites, it may be possible for *A. mellea* genets to become established within the bounds of *A. gallica* genets by colonizing stressed root systems before they become vulnerable to *A. gallica*. Mohammed and Guillaumin (1989) found that when *A. gallica* and *A. mellea* occupied the same substrate, *A. mellea* inhibited *A. gallica*, without being inhibited in return by *A. gallica*.

Collection of additional isolates (e.g., autumn 1996) will undoubtedly refine genet map precision, and may reveal the existence of previously undetected genets or even species (Watling 1995). We do not expect that *Armillaria* genet boundaries shift rapidly, especially along borders shared with other genets of the same species. Neither do we expect to witness a sudden abundance of *Armillaria* activity in the dry plot. The lower activity of *Armillaria* spp. in the dry plot may be due to: (1) the random failure of any *Armillaria* genets to establish over time in that portion of the landscape, (2) a less suitable distribution or lower volume of foodbase material to support rhizomorph growth and/or fruiting, or (3) long-term effects of the dryer nature of the dry plot. Wood decay and/or rhizomorph growth rate(s) may be reduced in scale and occur deeper in the soil profile on the dry plot than on the ambient and wet plots, thus perhaps also influencing fruiting at the soil surface of the dry plot. *Armillaria* spp. rhizomorph biomass, and rates of *Armillaria* spp. wood decay and rhizomorph production and growth at various depths in the TDE soils would be worth studying.

Regardless of the reasons for the current distributions of *A. gallica* and *A. mellea* genets in the TDE plots, these distributions have the potential to influence the expression of TDE treatment effects in ways that confound the experimental design. Knowledge of *Armillaria* genet distributions and their levels of activity will be important for the interpretation of TDE results. Though impractical for the TDE, substantial treatment replication along with due consideration for plot size would be helpful to statistically absorb the influences of *Armillaria* genets in field experiment designs.

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# THE GYPSY MOTH IN THE CENTRAL HARDWOODS:

# RESEARCH AND MANAGEMENT NEEDS

Robert Lawrence[1], Susan Burks[2], Dennis Haugen[3], and Marc Linit[4]

**Abstract:** The gypsy moth, *Lymantria dispar* (L.), is the most serious insect defoliator of trees in the Eastern United States. It is currently established in the area northeast of a line from Michigan to Virginia, and occupies most of the Adirondack and Laurentian Mixed Forest Provinces dominated by northern hardwood, spruce and fir forests. The range of the moth continues to expand to the west and south at a rate of 10-15 miles per year and is beginning to enter the Eastern Broadleaf Forest Province dominated by oak-hickory forests. Evaluation of potential impacts from gypsy moth defoliation on oak-hickory forests must now be based on studies conducted in the Eastern United States. Forest management strategies based on extrapolations from eastern forest types are unlikely to yield satisfactory results in the Central States.

The gypsy moth is expected to reach the Missouri Ozarks by the year 2025. Gypsy moth defoliation will be a new stress factor on the oak-hickory forests of Missouri. Periodic stresses due to drought, frost injury, poor soils, two-lined chestnut borer, and Armillaria root rot currently combine to cause tree decline and mortality. The added stress of gypsy moth defoliation is expected to be significant, although its magnitude is not known. Silvicultural techniques to minimize impacts by improving stand vigor and diversity need to be considered in the context of long term objectives for Ozark resources. Insufficient information is currently available to integrate these practices with proven techniques applicable to the Central Hardwoods.

The Steering Committee of the Missouri Cooperative Gypsy Moth Program is developing a Comprehensive Plan for Missouri. The plan aims to guide gypsy moth survey, detection and management decisions through the transition from an uninfested state to the time when the entire state is generally infested with the moth. The first priority is to delay the establishment of the gypsy moth in Missouri for as long as it is ecologically and economically feasible. The second priority is to define and implement research necessary to provide the basis for sound management decisions within our forest types and growing conditions.

[1] Forest Entomologist, Missouri Department of Conservation, P.O. Box 180, Jefferson City, MO 65102.
[2] Forest Pathologist, Missouri Department of Conservation, P.O. Box 180, Jefferson City, MO 65102.
[3] Forest Entomologist, USDA Forest Service, 1992 Folwell Avenue, St. Paul, MN 55108.
[4] Professor, Department of Entomology, University of Missouri, Columbia, MO 65211.

# SUMMARY OF MORTALITY STATISTICS AND FOREST HEALTH MONITORING RESULTS FOR THE NORTHEASTERN UNITED STATES

William H. McWilliams, Stanford L. Arner, and Charles J. Barnett[1]

Abstract: The USDA Forest Service's Forest Inventory and Analysis (FIA) program and the Forest Health Monitoring (FHM) program maintain networks of sample locations providing coarse-scale information that characterize general indicators of forest health. Tree mortality is the primary FIA variable for analyzing forest health. Recent FIA inventories of New York, Pennsylvania, and West Virginia reveal that the current rate of mortality, expressed as percent of basal area of live trees, is 1.01. The Virginia Pine (*Pinus virginiana*) and Pitch Pine (*Pinus rigida*) forest types, growing along the ridges of the Appalachian Mountains, had high rates of mortality compared to the average rate for the study region. By diameter class, mortality rates were above average in the two smallest diameter classes and the largest diameter class. Counties with above-average mortality were concentrated in northern New York, the southern half of Pennsylvania, and in a patchwork pattern in West Virginia. The top five mortality species/species groups were American elm (*Ulmus americana*), balsam fir (*Abies balsamea*), black locust (*Robinia pseudoacacia*), aspen (*Populus grandidentata* and *P. tremuloides*), and Virginia pine. The FHM program used crown density, foliage transparency, and crown dieback as indicators of forest health. The most current FHM results indicate that 97.3% of hardwood sample trees in New England and 97.5 in the three Mid-Atlantic states had crown density of greater than 20%. In terms of foliage transparency, 98.9% of the hardwood sample trees in New England had foliage transparency of 30% or less. For the Mid-Atlantic states, 92.5% had foliage transparency of 30% or less. Also, 95.8 and 98.1% of the hardwood sample trees had crown dieback of 20% or less in the New England and Mid-Atlantic regions, respectively.

## INTRODUCTION

Forest health has become a dominant issue within the forestry community. Although a firm consensus of what forest health is and how it should be measured is still evolving, it is important to seek sources of information to shed light on this important issue. The USDA Forest Service's Forest Inventory and Analysis (FIA) and Forest Health Monitoring (FHM) programs maintain networks of sample locations that provide coarse information that addresses general aspects of forest health. This information can help other researchers and policymakers in directing resources toward more detailed studies of species and regions that may be experiencing health-related problems. Because all states do not participate in the FHM program or have recent FIA inventories, it is our intent to combine the most recent findings of each program to characterize forest health in the northeastern United States.

## METHODS

### The FIA Program

The FIA program has been conducting successive forest inventories on a state-by-state basis since the 1930's. The northeastern FIA (NE-FIA) maintains a grid of roughly 16,000 sample locations across 13 states, from Ohio and West Virginia northward and eastward to Maine.

[1] Foresters, Northeastern Forest Experiment Station, USDA Forest Service, 5 Radnor Corporate Center, Suite 200, Radnor, PA 19087-4585.

Sample locations in each northeastern state are measured every 10 to 15 years using a two-phase sample design that comprises an aerial-photo interpretation phase and a ground-sample phase. In the photo-interpretation phase, large numbers of temporary photopoints are classified as forest and nonforest land. The intensity of photo sampling is high due to the low cost of aerial photo sampling relative to ground sampling. The photo sample is used in conjunction with U.S. Bureau of Census data on land area to produce estimates of forest land. In addition to classifying land use at each photopoint, a volume-per-acre class is assigned. Each ground-sample location is photointerpreted in the same manner as the photopoints. The ground locations are then chosen for measurement in proportion to the distribution of photopoints by volume-per-acre class.

Ground locations are visited to verify photo interpretation and to collect a suite of location- and tree-level measurements. Over the years, the northeastern FIA has used a variety of variable- and fixed-radius plot designs. The set of sample locations includes both remeasured locations and new ground locations. Among the location variables are slope, aspect, physiographic class, and stand origin. Tree data include variables such as species, diameter at breast height (dbh), tree history (live, cut, dead), bole length, and tree grade. Tree measurements are used to compile additional location variables such as forest type, stand size, and stand density. Other derived variables include growing-stock volume, net growth, removals, and mortality. Expansion factors for individual trees and sample locations (derived from the aerial-photo sample) are then used to compile population estimates for the state of interest. Each sample location represents roughly 5,000 to 10,000 acres of forest land.

Three of NE-FIA's most recent inventories are for Pennsylvania (Alerich 1993), New York (Alerich and Drake 1995), and West Virginia (DiGiovanni 1990). Tree mortality is the primary quantitative measure we have for assessing forest health using FIA data. Mortality estimates are computed using remeasured sample locations. The estimates represent changes that have occurred between the two most recent inventories. The following is sample information for the three Mid-Atlantic states:

| State | Inventory period | Number of sample locations | Number of sample trees |
|---|---|---|---|
| New York | 1980 to 1992 | 2,457 | 39,134 |
| Pennsylvania | 1978 to 1988 | 1,938 | 33,365 |
| West Virginia | 1975 to 1988 | 1,253 | 14,785 |

Mortality is expressed on an annual basis to provide information that can be compared among states, regions, species, and forest types. For this study, mortality estimates were indexed as a percentage of total basal area, numbers of trees, and growing-stock volume. Because we lack digital data for earlier inventories, we were unable to compile longer term trend information on mortality. Also, the sample is limited to trees that were at least 5.0 inches in dbh at the time of the previous inventory because estimates for smaller trees are not based on paired measurements.

The FHM Program

The FHM program first established plots in 1990 in the six New England states. As of 1995, the northern FHM program had 1,294 plots in 14 states, 672 of which were classified as forested. Currently, the northern states are grouped into three broad geographic regions: New England (Connecticut, Maine, Massachusetts, New Hampshire, Rhode Island, and Vermont); Mid-Atlantic (Delaware, Maryland, New Jersey, Pennsylvania, and West Virginia); and Lake States (Michigan, Minnesota, and Wisconsin). Plots were established in Delaware, Maryland, and New Jersey in 1991; in Michigan, Minnesota, and Wisconsin in 1994; and in Pennsylvania and West Virginia in 1995. Results from plots established in the New England and Mid-Atlantic regions are discussed here.

The FHM program, which combines research by the USDA Forest Service, state agencies, and other federal agencies, began as a part of the Environmental Monitoring and Assessment Program (EMAP) of the U.S. Environmental Protection Agency. EMAP developed a systematic sampling grid that covers North America. A sample location associated with each point on this grid is classified as forested or not forested. If a location is not forested, it is maintained in the sample and revisited every 4 years to determine whether it has become forested. If a plot is

forested, location- and tree-level data are collected. Each forested location is a cluster of four fixed-radius, 1/24-acre circular subplots spaced 120 feet apart in a triangular formation (one center subplot with three subplots at the vertices of an equilateral triangle centered around the first subplot).[2]

When a sample location is established, location- and tree-level data are collected. Location data include location and physiographic information, understory vegetation assessments, and land-use information. Tree-level data include species, dbh, and assessments of crown condition, such as crown dieback, foliage transparency, and crown density. Location information and some tree data (e.g., dbh) are updated every 4 years. Data on crown condition are collected annually on forested plots.

The definitions of the three crown variables included in this report are:

Crown Dieback: Branch mortality that begins at the terminal portion of a branch and proceeds toward the trunk and/or base of the live crown.

Foliage Transparency: Amount of skylight visible through the live, normally foliated portion of the crown or branch.

Crown Density: Amount of crown branches, foliage, and reproductive structures that block visible light through the crown (Tallent-Halsell 1994).

Data for these variables are recorded in 5% classes from 0 to 100 (0-5, 6-10, etc.). Data are grouped for each variable and reported here as the percentage of trees in each group.

## RESULTS

### Recent FIA Inventories

The average rates of mortality for all states combined were 1.01, 1.21, and 0.75% for basal area, numbers of trees, and growing-stock volume, respectively (Table 1). Because longer term trend data for mortality are unavailable, it is difficult to qualify estimates as high or low. The overall averages provide benchmarks to put mortality estimates for specific classes and species into perspective. Usually, the rate expressed as numbers of trees is higher than the others because mortality often is concentrated in smaller size classes that contain more trees. The rate expressed in growing-stock volume tends to be lower because it excludes rough and rotten trees, which tend to die more often than growing-stock trees due to preexisting conditions such as butt rot. The rate based on basal area probably is the most useful because it accounts for all trees that were alive at the previous inventory and is not limited to trees selected for management (growing stock). Mortality rates were slightly higher than the overall averages in Pennsylvania. Mortality rates in New York and West Virginia were below the averages and similar in magnitude.

Results by forest-type group reflect species composition of the group, site-species relationships, and dynamic factors affecting the site, such as weather, insects, disease, and anthropogenic influences. Mortality averaged 1.01% using basal area as a measure and ranged from 1.62% for the Loblolly-Shortleaf group to 0.74% for the Red-White-Jack Pine group (Table 2). Loblolly-Shortleaf stands in the Mid-Atlantic states comprise primarily the Virginia Pine (*Pinus virginiana* Mill.) and Pitch Pine (*Pinus rigida* Mill.) forest types, as well as some Eastern Redcedar (*Juniperus virginiana* L.) and Table-Mountain Pine (*Pinus pungens* Lamb). The Virginia Pine and Pitch Pine types often occupy dry, rocky sites of the Appalachian Mountains. Common associates include a variety of oaks (*Quercus* spp.), red maple (*Acer rubrum* L.), blackgum (*Nyssa sylvatica* Marsh.), and miscellaneous species.

[2] Conkling, B.L. and G.E. Byers., eds. 1992. Forest health monitoring field methods guide (national guide). Unpublished report on file at U.S. Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC. 434 p.

Table 1. Average annual mortality rate and standard error (SE) expressed as percent of basal area of live trees, numbers of live trees, and growing-stock volume, New York 1980-1992, Pennsylvania 1978-1988, and West Virginia 1975-1988

| State | Basal area | | No. of trees | | Growing-stock volume | |
|---|---|---|---|---|---|---|
| | Estimate | SE | Estimate | SE | Estimate | SE |
| New York | 0.96 | .02 | 1.11 | .02 | 0.69 | .02 |
| Pennsylvania | 1.07 | .03 | 1.30 | .03 | 0.84 | .03 |
| West Virginia | 0.98 | .03 | 1.23 | .03 | 0.68 | .03 |
| All states combined | 1.01 | .01 | 1.21 | .01 | 0.75 | .01 |

Table 2. Average annual mortality rate and standard error (SE) expressed as percent of basal area of live trees, numbers of live trees, and growing-stock volume by forest-type group, New York 1980-1992, Pennsylvania 1978-1988, and West Virginia 1975-1988

| Forest-type group | Basal area | | No. of trees | | Growing-stock volume | |
|---|---|---|---|---|---|---|
| | Estimate | SE | Estimate | SE | Estimate | SE |
| Loblolly-Shortleaf[1] | 1.62 | .19 | 1.72 | .23 | 1.35 | .17 |
| Aspen-Birch | 1.32 | .13 | 1.55 | .15 | 1.11 | .12 |
| Elm-Ash-Maple | 1.29 | .09 | 1.47 | .10 | 1.04 | .09 |
| Spruce-Fir | 1.23 | .11 | 1.35 | .14 | 1.10 | .11 |
| Oak-Hickory | 1.07 | .02 | 1.31 | .03 | 0.81 | .03 |
| Oak-Pine | 1.01 | .09 | 1.21 | .10 | 0.82 | .10 |
| Maple-Beech-Birch | 0.94 | .02 | 1.10 | .02 | 0.66 | .02 |
| Miscellaneous | 0.80 | .26 | 1.08 | .38 | 0.89 | .51 |
| Red-White-Jack pine | 0.74 | .04 | 0.97 | .05 | 0.55 | .03 |
| All groups combined | 1.01 | .01 | 1.21 | .01 | 0.75 | .01 |

[1]Includes Virginia Pine, Pitch Pine, Table-Mountain Pine, and Eastern Redcedar forest types.

Throughout the infestation area of gypsy moth (*Lymantria dispar* L.), along the ridges of the Appalachian Mountains, the Virginia Pine and Pitch Pine communities were particularly susceptible to defoliation and subsequent mortality. The Aspen-Birch, Elm-Ash-Maple, and Spruce-Fir groups also had high mortality rates. The common thread among these groups is that they have characteristic species that are experiencing high mortality, such as aspen, American elm, and balsam fir.

Stocking often is mentioned as a possible factor that influences tree mortality. The results of this study show that mortality rates were similar by stocking class (Table 3). Rates for stands in the 140% and higher stocking class were roughly one-third to one-half higher than the overall averages, but were associated with high standard errors. Rates were slightly lower in the most sparsely stocked (0 to 19% stocked) stands.

The overall averages allow us to focus on mortality rates that are "above average" and that may indicate prospective issues related to forest health. The average mortality rate as a percentage of live tree basal area was 1.01%. These data can be displayed by county to provide a general idea of the spatial distribution of mortality (Figure 1a). Counties with above-average mortality were concentrated in northern New York and southern Pennsylvania. Above-average mortality was scattered in West Virginia.

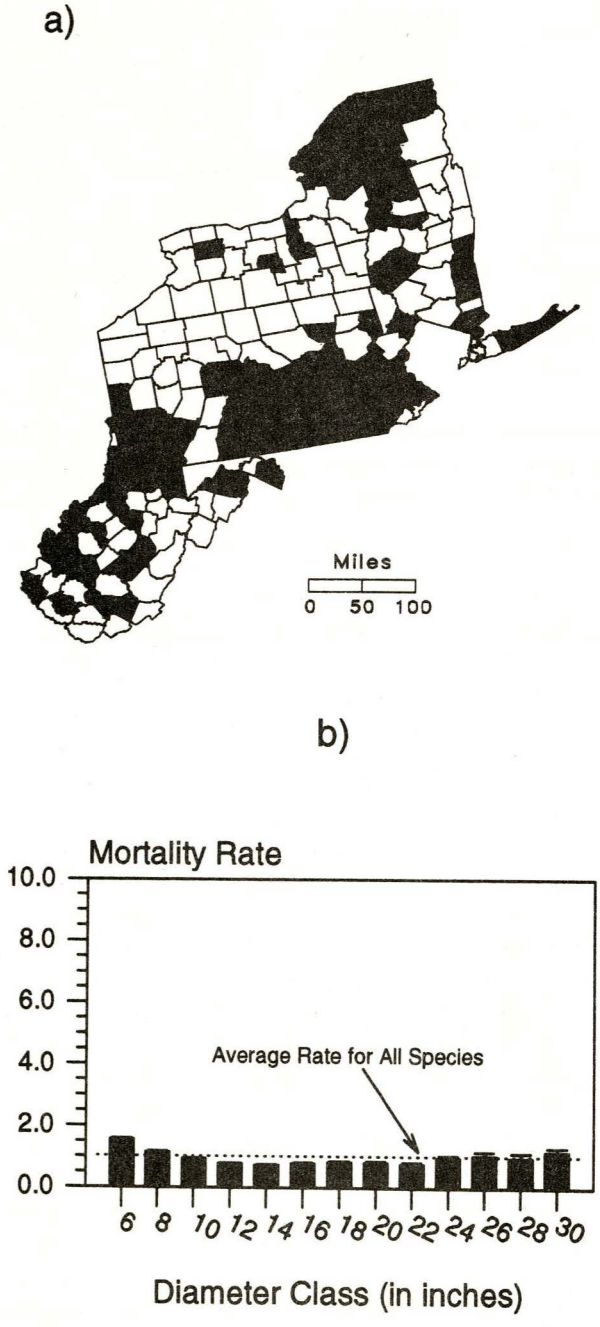


Figure 1. ( a) Counties with above-average annual rates of mortality as percentage of all species/species groups basal area of live trees (5.0 inches in diameter and larger); and (b) Average annual rate of mortality of all species/species groups live tree basal area by diameter class where brackets indicate one standard error (standard errors less than 0.1 are obscured).

Table 3. Average annual mortality rate and standard error (SE) expressed as percent of basal area of live trees, numbers of live trees, and growing-stock volume by stocking class, New York 1980-1992, Pennsylvania 1978-1988, and West Virginia 1975-1988

| Stocking Class (percent) | Basal area Estimate | SE | No. of trees Estimate | SE | Growing-stock volume Estimate | SE |
|---|---|---|---|---|---|---|
| 0 to 19 | 0.88 | .06 | 0.99 | .08 | 0.71 | .06 |
| 20 to 39 | 0.98 | .04 | 1.14 | .05 | 0.74 | .04 |
| 40 to 59 | 0.99 | .03 | 1.17 | .03 | 0.71 | .03 |
| 60 to 79 | 0.98 | .02 | 1.20 | .03 | 0.73 | .02 |
| 80 to 99 | 1.08 | .03 | 1.33 | .04 | 0.82 | .03 |
| 100 to 119 | 0.99 | .05 | 1.25 | .06 | 0.73 | .05 |
| 120 to 139 | 0.89 | .09 | 1.15 | .12 | 0.54 | .09 |
| 140 and higher | 1.34 | .40 | 1.31 | .30 | 1.16 | .40 |
| All classes combined | 1.01 | .01 | 1.21 | .01 | 0.75 | .01 |

Table 4. Average annual mortality rate and standard error (SE) expressed as percent of basal area of live trees, numbers of live trees, and growing-stock volume by species/species group for species with above average mortality, New York 1980-1992, Pennsylvania 1978-1988, and West Virginia 1975-1988

| Species/Species Group | Basal area Estimate | SE | No. of trees Estimate | SE | Growing-stock volume Estimate | SE |
|---|---|---|---|---|---|---|
| American elm | 3.28 | .20 | 3.26 | .22 | 3.22 | .22 |
| Balsam fir | 2.64 | .23 | 2.38 | .21 | 2.71 | .26 |
| Black locust | 2.40 | .14 | 2.75 | .15 | 1.61 | .15 |
| Aspen | 2.22 | .10 | 2.45 | .11 | 1.91 | .10 |
| Virginia pine | 1.96 | .15 | 1.97 | .17 | 1.87 | .16 |
| Birch | 1.37 | .05 | 1.61 | .06 | 1.00 | .05 |
| Chestnut Oak | 1.34 | .07 | 1.56 | .08 | 1.18 | .07 |
| Black Walnut | 1.24 | .18 | 1.19 | .20 | 0.87 | .19 |
| Red spruce | 1.16 | .09 | 1.36 | .12 | 1.07 | .09 |
| American beech | 1.09 | .05 | 0.89 | .05 | 0.64 | .05 |

The distribution of mortality by diameter class (Figure 1b) revealed above-average rates in the two smallest diameter classes and the largest diameter class. This fits with the notion that as northeastern forests mature, smaller trees are being crowded out. Also, large "older" trees may be dying as middle-aged trees compete for growing space.

Ten species/species groups out of 23 were identified as having above average mortality (Table 4). Five species had roughly double the average rate of mortality. American elm (*Ulmus americana* L.) had the highest rate at 3.28, or more than triple the average rate. The impact of a wilt fungus (*Ceratocystis ulmi* (Buisum.) C. Moreau), the agent of Dutch elm disease, has been well documented since its introduction in 1930 (Burns and Honkala 1990b). Concentrations of dead American elm trees were highest in eastern New York (Figure 2a). Other than a high rate of mortality in some large diameter classes, mortality rates were similar in other size classes (Figure 2b).

Balsam fir (*Abies balsamea* (L.) Mill.) ranked second in mortality with a rate of 2.64%. Spruce budworm (*Choristoneura fumiferana* (Clemens)), the primary damaging agent of balsam fir, thrives in stands with a high proportion of fir. Most of the mortality in the Mid-Atlantic states was in northeastern New York, where balsam fir is abundant (Figure 3a). Larger trees in the sample had the highest mortality rates (Figure 3b).

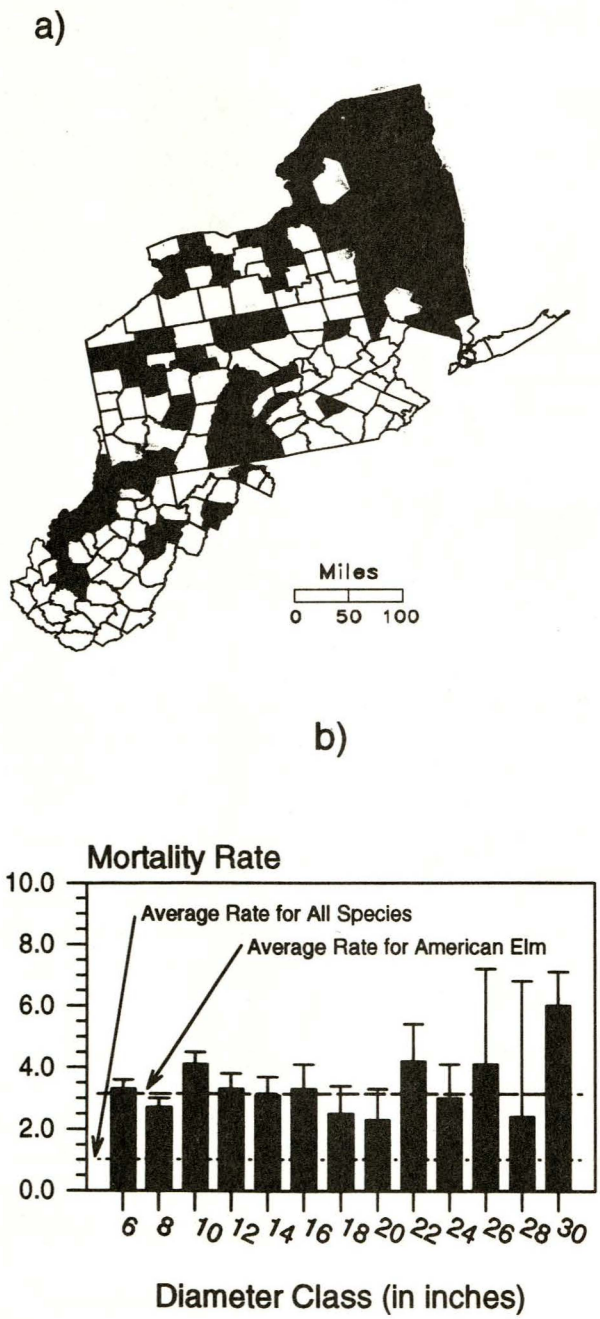


Figure 2. (a) Counties with above-average annual rates of mortality as percentage of American elm basal area of live trees (5.0 inches in diameter and larger); and (b) Average annual rate of mortality of American elm live-tree basal area by diameter class (brackets indicate one standard error).

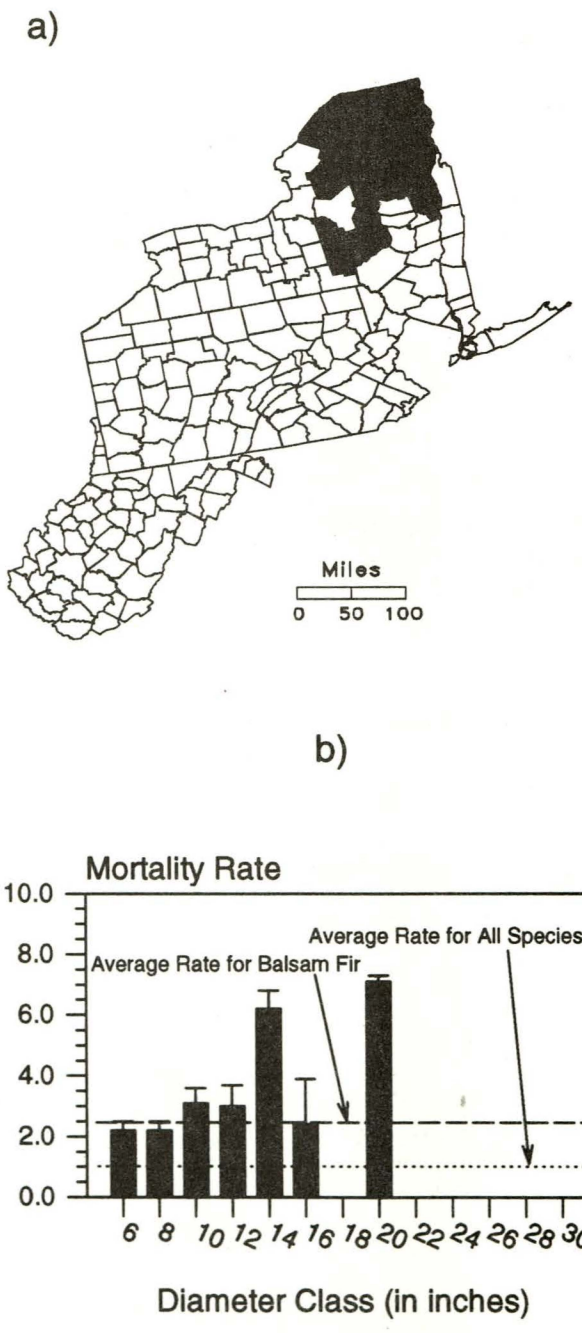


Figure 3. (a) Counties with above-average annual rates of mortality as percentage of balsam fir basal area of live trees (5.0 inches in diameter and larger); and (b) Average annual rate of mortality of balsam fir live-tree basal area by diameter class (brackets indicate one standard error).

Black locust (*Robinia pseudoacacia*) had the third highest mortality rate (2.40%). Areas with above average mortality were concentrated in West Virginia and southwestern Pennsylvania (Figure 4a). By diameter class, mortality rates were highest in the smallest diameter class (Figure 4b). Black locust is a pioneer species that is relatively short lived in the development of a typical stand (Burns and Honkala 1990b). Also, black locust usually is a small component in forest stands. It accounts for less than 1% of the growing-stock volume in the study region.

Aspen (*Populus grandidentata* Michx. and *P. tremuloides* Michx.) had the fourth highest mortality. Aspen is another species whose relatively high mortality rate (2.22%) is surprising. Aspen mortality was prevalent in counties where it is most common, primarily in Pennsylvania and New York (Figure 5a). Mortality was average or below average in all but the smallest and largest diameter classes (Figure 5b). Aspen mortality in the largest class was roughly twice the average rate for that species, possibly due to poor vigor in relation to other species in stands in advanced stages of development. Aspen is highly intolerant of shade and relatively short lived.

Virginia pine ranked fifth in mortality with a rate of 1.96% with most of the mortality in West Virginia and some mountainous counties in Pennsylvania (Figure 6a). Mortality of Virginia pine was spread relatively evenly over the range of diameter classes (Figure 6b). A pioneer species that regenerates on mineral soil and is intolerant of shade, Virginia pine often is replaced by deciduous species as a normal part of stand development (Burns and Honkala 1990a).

Species with mortality rates higher than the overall average (at least 30% higher) were the birches (*Betula* spp.) and chestnut oak (*Quercus prinus* L.) (Table 4). Other species with above-average rates are black walnut (*Juglans nigra* L.), red spruce (*Picea rubens* Sarg.), and American beech (*Fagus grandifolia* Ehrh.).

Recent FHM Inventories

The FHM program has 480 sample locations in 11 states in New England and the Mid-Atlantic states. Of these, 249 are forested on all or part of the sample. In New England, 79% of the area covered by the 263 locations is forested. In the Mid-Atlantic states, 51% of the area covered by 217 locations is forested.

In 1995, there were 6,020 live trees larger than 5.0 inches dbh on the New England FHM sample locations. Of these, 3,375 were hardwood species (Table 5). Red maple, sugar maple (*Acer saccharum* Marsh.), and paper birch (*Betula papyrifera* Marsh.) made up 52% of the hardwood sample in New England. Yellow birch (*Betula alleghaniensis* Britton), American beech, aspen, northern red oak (*Quercus rubra* L.), and white/green ash (*Fraxinus americana* L. and *F. pennsylvanica* Marsh.) made up another 33% of the sample. More than 90% of the trees in the white/green ash group were white ash.

There were 2,460 live trees on the FHM sample locations in the mid-Atlantic states, 2,275 of which were hardwood species. Red maple, white oak, black cherry (*Prunus serotina* Ehrh.), sugar maple, northern red oak, various hickory species (*Carya* spp.), American beech, and sweetgum (*Liquidambar styraciflua* L.) made up 54% of the sample.

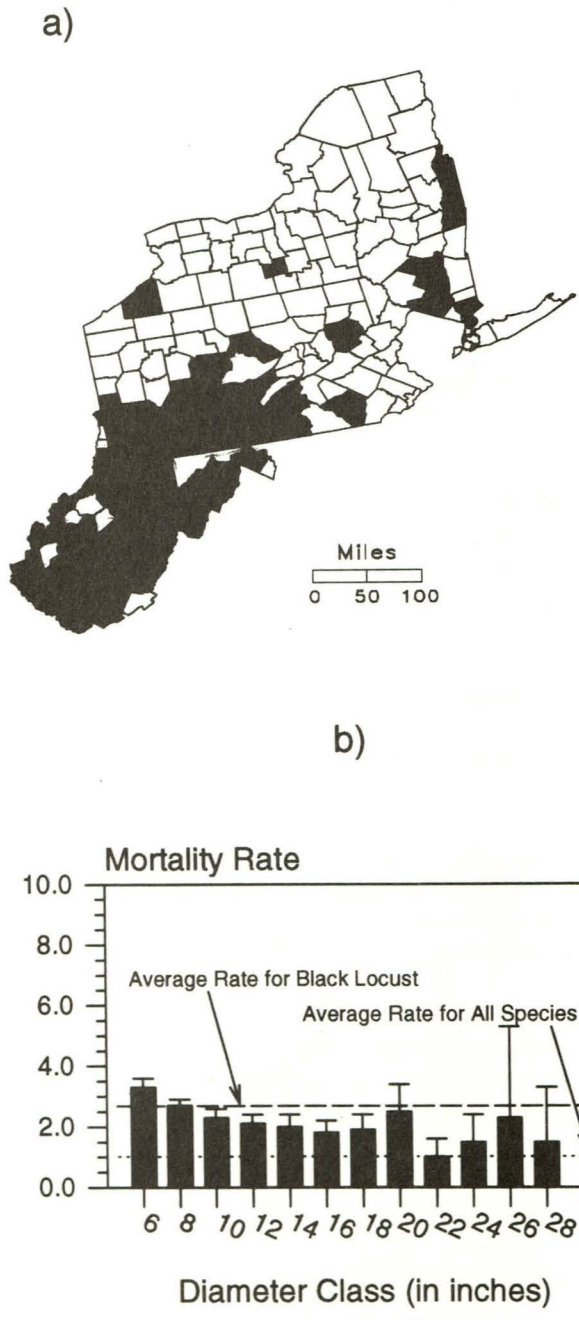


**Figure 4.** (a) Counties with above-average annual rates of mortality as percentage of black locust basal area of live trees (5.0 inches in diameter and larger); and (b) Average annual rate of mortality of black locust live-tree basal area by diameter class (brackets indicate one standard error).

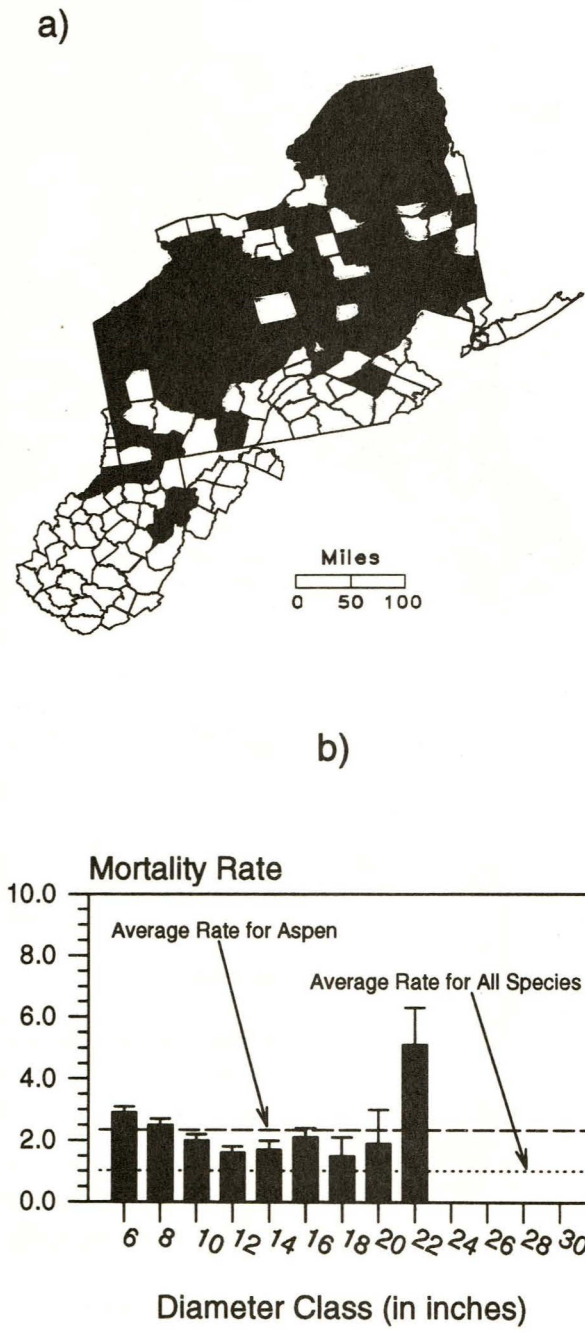


Figure 5. (a) Counties with above-average annual rates of mortality as percentage of aspen basal area of live trees (5.0 inches in diameter and larger); and (b) Average annual rate of mortality of aspen live-tree basal area by diameter class (brackets indicate one standard error).

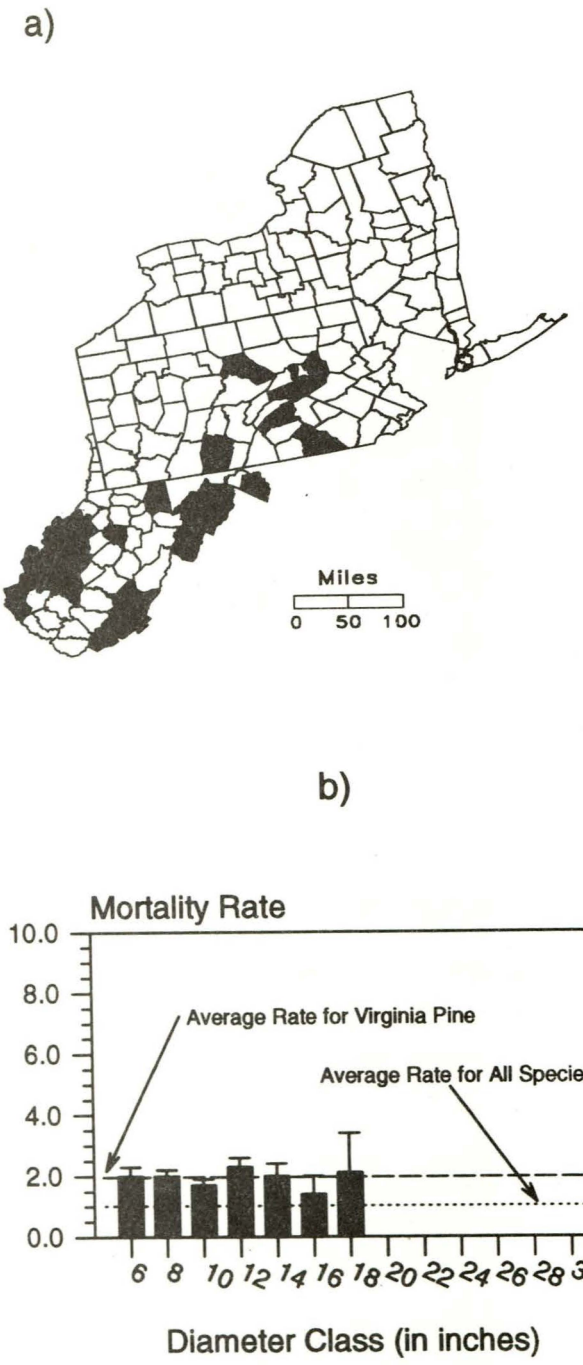


Figure 6. (a) Counties with above-average annual rates of mortality as percentage of Virginia pine basal area of live trees (5.0 inches in diameter and larger), and (b) Average annual rate of mortality of Virginia pine live-tree basal area by diameter class (brackets indicate one standard error).

Table 5. Distribution of live trees of major hardwood species on FHM sample locations in New England and Mid-Atlantic states by crown-density class[3]

| Species | Sample size | Crown density: Good (51+%) | Average (21-50%) | Poor (1-20%) |
|---|---|---|---|---|
| | Number | Percent of sampled trees | | |
| | | New England | | |
| Red maple | 991 | 30.4 | 66.4 | 3.2 |
| Sugar maple | 448 | 48.4 | 50.2 | 1.3 |
| Paper birch | 305 | 41.6 | 56.7 | 1.6 |
| Yellow birch | 267 | 52.4 | 46.1 | 1.5 |
| American beech | 249 | 29.3 | 67.1 | 3.6 |
| Aspen | 221 | 26.7 | 66.5 | 6.8 |
| Northern red oak | 213 | 38.5 | 60.1 | 1.4 |
| White/green ash | 182 | 39.0 | 56.6 | 4.4 |
| Softwoods | 2,645 | 33.4 | 64.3 | 2.3 |
| Hardwoods | 3,375 | 38.8 | 58.5 | 2.7 |
| All species | 6,020 | 36.4 | 61.1 | 2.5 |
| | | Mid-Atlantic | | |
| Red maple | 414 | 56.8 | 39.1 | 4.1 |
| White oak | 193 | 53.4 | 44.6 | 2.1 |
| Black cherry | 128 | 52.3 | 44.5 | 3.1 |
| Sugar maple | 127 | 63.8 | 35.4 | 0.8 |
| Northern red oak | 112 | 63.4 | 36.6 | 0.0 |
| Hickory | 104 | 75.0 | 25.0 | 0.0 |
| American beech | 73 | 63.0 | 35.6 | 1.4 |
| Sweetgum | 62 | 59.7 | 40.3 | 0.0 |
| Softwoods | 203 | 35.5 | 63.1 | 1.5 |
| Hardwoods | 2,257 | 60.2 | 37.3 | 2.5 |
| All species | 2,460 | 58.2 | 39.4 | 2.4 |

Crown Density. Approximately 97.3% of hardwood trees in New England had crown density of greater than 20% (Table 5). These percentages were consistent with results from previous years. For both aspen and white/green ash, more than 4% of the trees had crown densities of 20% or less.

[3] Barnett, C.J.; Mielke M.; Twardus, D.; Miller-Weeks, M. Northern forest health monitoring: 1995 summary reports. USDA, Forest Service, Northeastern Forest Experiment Station and Northern Area, State and Private Forestry. In prep.

Table 6. Distribution of live trees of major hardwood species on FHM sample locations in New England and Mid-Atlantic states by foliage-transparency class[3]

| Region/Species | Sample size | Foliage transparency: Normal (0-30%) | Moderate (31-50%) | Severe (51+%) |
|---|---|---|---|---|
| | Number | Percent of sampled trees | | |
| | | New England | | |
| Red maple | 991 | 99.1 | 0.4 | 0.5 |
| Sugar maple | 448 | 99.8 | 0.0 | 0.2 |
| Paper birch | 305 | 97.7 | 1.6 | 0.7 |
| Yellow birch | 267 | 99.3 | 0.0 | 0.7 |
| American beech | 249 | 99.6 | 0.4 | 0.0 |
| Aspen | 221 | 98.2 | 1.8 | 0.0 |
| Northern red oak | 213 | 99.5 | 0.0 | 0.5 |
| White/green ash | 182 | 97.8 | 0.5 | 1.6 |
| Softwoods | 2,645 | 99.0 | 0.9 | 0.1 |
| Hardwoods | 3,375 | 98.9 | 0.6 | 0.5 |
| All species | 6,020 | 99.0 | 0.7 | 0.3 |
| | | Mid-Atlantic | | |
| Red maple | 414 | 93.2 | 4.4 | 2.4 |
| White oak | 193 | 95.3 | 3.1 | 1.6 |
| Black cherry | 128 | 77.3 | 18.0 | 4.7 |
| Sugar maple | 123 | 96.9 | 1.6 | 1.6 |
| Northern red oak | 112 | 92.9 | 7.1 | 0.0 |
| Hickory | 104 | 99.0 | 1.0 | 0.0 |
| American beech | 73 | 94.5 | 4.1 | 1.4 |
| Sweetgum | 62 | 100.0 | 0.0 | 0.0 |
| Softwoods | 203 | 90.6 | 8.4 | 1.0 |
| Hardwoods | 2,257 | 92.5 | 5.8 | 1.7 |
| All species | 2,460 | 92.3 | 6.1 | 1.6 |

In the mid-Atlantic region, 97.5% of hardwood trees had crown density of greater than 20%. Red maple had 4.1% of trees in the 20% or less class. Black cherry was the only other species with more than the 2.4% average for crown density of 20% or less.

Foliage transparency. In New England, 98.9% of hardwood trees were classified as having less than 30% foliage transparency (Table 6), compared to 98.7 and 98.9% in 1993 and 1994, respectively. Trees in the white/green ash group had three times the average percentage of trees in the greater than 50% transparency class. Aspen had three times the average percentage of trees in the moderate transparency class.

For Mid-Atlantic states, 92.3% of all the trees had foliage transparency of less than 30% in 1995, compared to 97.3% in 1994. The number of sample locations in this region increased significantly in 1995 by the addition of plots in Pennsylvania and West Virginia. Black cherry had the highest levels of foliage transparency with 18.0% in the 31-50% class and 4.7% in the greater than 50% class. Northern red oak also had a higher than average percentage of trees in the 31-50% class.

Crown Dieback. In New England, 95.8% of the hardwood trees on FHM plots were classified as having crown dieback of 20% or less (Table 7). Hardwoods had higher percentages in the 6-20%, 21-50%, and greater than 50% classes than did softwoods. The percentage of white and green ash trees in the 21-50% and greater than 50% classes decreased from 10.2% in 1994 to 8.8% in 1995. There were 8.6% of the aspen trees in the 21-50% and greater than 50% classes. American beech had a slightly higher than the average percentage of trees in the 21-50% and greater than 50% classes.

In the Mid-Atlantic states, 98.1% of the hardwood trees had crown dieback 20% or less. No single species had more than 3.0% of trees in the moderate class. However, northern red oak had nearly double the average percentage of trees in the light and moderate classes, with 12.5 and 1.8%, respectively.

## DISCUSSION

The FIA and FHM programs are providing valuable information on forest health. The FIA mortality data provide estimates of overall averages that allow comparisons among classes of forest and tree species, as well as coarse spatial information. Still, it is difficult to qualify mortality rates as high or low because there is little published information on this topic. In the northeast, we are further constrained by not having digital data for more than the current inventory. And the FIA data have other limitations. Because dead trees are relatively rare within the FIA sample compared to live "survivor" trees, mortality estimates have relatively high sampling errors. Also, it can be difficult to distinguish tree removal from mortality in areas where salvage cuttings occur. If an FIA field crew encounters a tree stump, the tree often is classified as cut even though it may have died before removal. Another limitation is that FIA data usually are insufficient to establish relationships of cause and effect. The reason for this is that FIA typically measures effects, or the dependent variable, rather than causal factors, or independent variables (Schreuder and Thomas 1991). Additional research is needed to determine the kinds of variables FIA could collect that would elucidate causation and that are economically feasible. The lag between the time when damage agents attack a tree and the time when the tree dies also should be considered in our analyses. For example, during the recent inventory of Pennsylvania, large-scale outbreaks of insects were occurring in northwestern Pennsylvania (McWilliams and others 1996), but the trees were still alive. We know that since then, significant numbers of trees have died. Lastly, it is important to note that using mortality rates as a measure of forest health ignores the role of regeneration in replenishing forest stands.

The FHM sample results suggest that tree health is good in New England and the Mid-Atlantic states. However in New England, aspen had higher than average percentages of trees in the poor-density and moderate-transparency classes. Also in 1995, there were reports of increased activity by the forest tent caterpillar (*Malacosoma disstria* Hubner) in the aspen-birch forest types.[3] Defoliation by forest tent caterpillar would reduce the density and increase transparency of the crowns of these trees. In New England, ash yellows continues to contribute to thin and dying crowns (i.e., low density, high transparency, high dieback) on these trees. Beech bark disease (*Nectria* spp.) is contributing to the higher than average dieback ratings on American beech. In the Mid-Atlantic states, the cherry scallop shell moth (*Calocalpe undulata* L.) likely contributed to the higher density and transparency ratings in black cherry. The elevated transparency and dieback in northern red oak probably was the result of scattered defoliation by gypsy moth and forest tent caterpillar.

Table 7. Distribution of live trees of major hardwood species on FHM sample locations in New England and Mid-Atlantic states by crown-dieback class[3]

| Region/Species | Sample size | Crown dieback: None (0-5%) | Light (6-20%) | Moderate (21-50%) | Severe (51+%) |
|---|---|---|---|---|---|
| | Number | ----- Percent of sampled trees ----- | | | |
| | | New England | | | |
| Red maple | 991 | 75.1 | 20.4 | 3.4 | 1.1 |
| Sugar maple | 448 | 84.2 | 12.7 | 1.8 | 1.3 |
| Paper birch | 305 | 79.0 | 17.1 | 2.0 | 2.0 |
| Yellow birch | 267 | 83.5 | 14.6 | 1.1 | 0.8 |
| American beech | 249 | 68.3 | 26.5 | 4.0 | 1.2 |
| Aspen | 221 | 64.7 | 26.7 | 7.2 | 1.4 |
| Northern red oak | 213 | 72.8 | 26.3 | 1.0 | 0.0 |
| White/green ash | 182 | 72.0 | 19.2 | 4.4 | 4.4 |
| Softwoods | 2,645 | 89.1 | 9.1 | 1.3 | 0.5 |
| Hardwoods | 3,375 | 76.4 | 19.4 | 2.9 | 1.3 |
| All species | 6,020 | 82.0 | 14.9 | 2.2 | 0.9 |
| | | Mid-Atlantic | | | |
| Red maple | 414 | 90.6 | 7.3 | 1.5 | 0.7 |
| White oak | 193 | 95.9 | 2.6 | 0.5 | 1.0 |
| Black cherry | 128 | 92.2 | 7.8 | 0.0 | 0.0 |
| Sugar maple | 127 | 98.4 | 1.6 | 0.0 | 0.0 |
| Northern red oak | 112 | 85.7 | 12.5 | 1.8 | 0.0 |
| Hickory | 104 | 94.2 | 5.8 | 0.0 | 0.0 |
| American beech | 73 | 90.4 | 6.9 | 1.4 | 1.4 |
| Sweetgum | 62 | 95.2 | 3.2 | 1.6 | 0.0 |
| Softwoods | 203 | 97.0 | 3.0 | 0.0 | 0.0 |
| Hardwoods | 2,257 | 91.8 | 6.3 | 1.1 | 0.9 |
| All species | 2,460 | 92.2 | 6.0 | 1.0 | 0.8 |

On the basis of assessments of trees within the FHM sample network in the northeast, the hardwood resource seems to be doing well, with noted exceptions. Because the network was able to identify species with known problems or conditions, we expect that changing conditions in other species can also be identified.

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# TRENDS IN HARDWOOD TIMBER RESOURCES FOR THE NORTHERN STATES

Richard H. Widmann[1] and Thomas L. Schmidt[2]

Abstract: The forests of the 20-state Northern Region are maturing. The portion of timberland covered by sawtimber-size stands has increased and the area in the sapling/seedling-size stands has decreased. Growing-stock volume has doubled since the early 1950's. Most of the recent gains in volume have occurred on sawtimber-size trees, resulting in large increases in sawtimber volume. Individual species are growing and being removed at different rates, causing changes in the species composition of the forest. For example, red maple (*Acer rubrum* L.) is increasing at a faster rate than many of the more desirable species, such as oaks (*Quercus* spp.).

## INTRODUCTION

The hardwood forest resource of the 20-state Northern Region of the United States (Figure 1) has continued to mature following intensive harvesting and land-clearing activities during the late 1800's and early 1900's[3]. Maturing of the forest has brought about changes in the structure, size, value, species composition, and use of the forest. With the maturation of the resource come additional opportunities to influence change.

Forest inventories conducted by the USDA Forest Service in the study region have tracked the maturing of the hardwood resource. In this paper we present the most current statistical data compiled by the Northeastern and North Central Forest Experiment Stations' Forest Inventory and Analysis (FIA) units, and describe trends in the hardwood timber resource in the Northern Region.

Sources for the data used in this report are the individual state resource reports published by FIA units (see References section) as well as a national assessment of timber supply (Powell and others 1993). The date for each inventory varies from state to state, as do the number of years between inventories. To identify trends, comparisons are made between the most current and previous inventories. The inventory cycle for the region is about 14 years, though this interval is shorter for many states with large hardwood resources.

Although we know that all types of forest produce important benefits, categorizing forest land is helpful in organizing forest management. Forest land is categorized by the USDA Forest Service as noncommercial; which includes reserved forest land, unproductive forest land and urban forests; or as timberland. Reserved forest land is unavailable for harvest due to administrative or legislative restrictions. Unproductive forest land is not capable of growing 20 cubic feet of wood annually when managed, which is the minimum growth necessary to qualify as timberland. Urban forests are removed from the timber resource base because of their location in urban and community areas and thus are unlikely to be managed for timber products. Timberland, previously referred to as commercial forest land, is physically capable of growing timber crops and is potentially available for harvesting.

While all forest lands produce multiple benefits, timberland is the resource base upon which a large and growing timber industry relies to maintain it's product flow. The future area of forest land in the noncommercial category is expected to increase over time as more and more forest lands are "set aside" for specific uses such as parks and

[1] Forester, Northeastern Forest Experiment Station, USDA Forest Service, 5 Radnor Corp. Ctr. Suite 200, Radnor, PA 19087-4585.

[2] Research Forester, North Central Forest Experiment Station, USDA Forest Service, 1992 Folwell Ave., St. Paul, MN 55108.

[3] The 20 state Northern Region comprised of the Northeast and North Central subregions is defined by the Resource Planning Act (RPA) Assessment Update, Forest Resources of the United States, 1992 (Powell and others 1993). Although Ohio is inventoried by the inventory work unit at the Northeastern Forest Experiment Station, it is included in the North Central subregion.

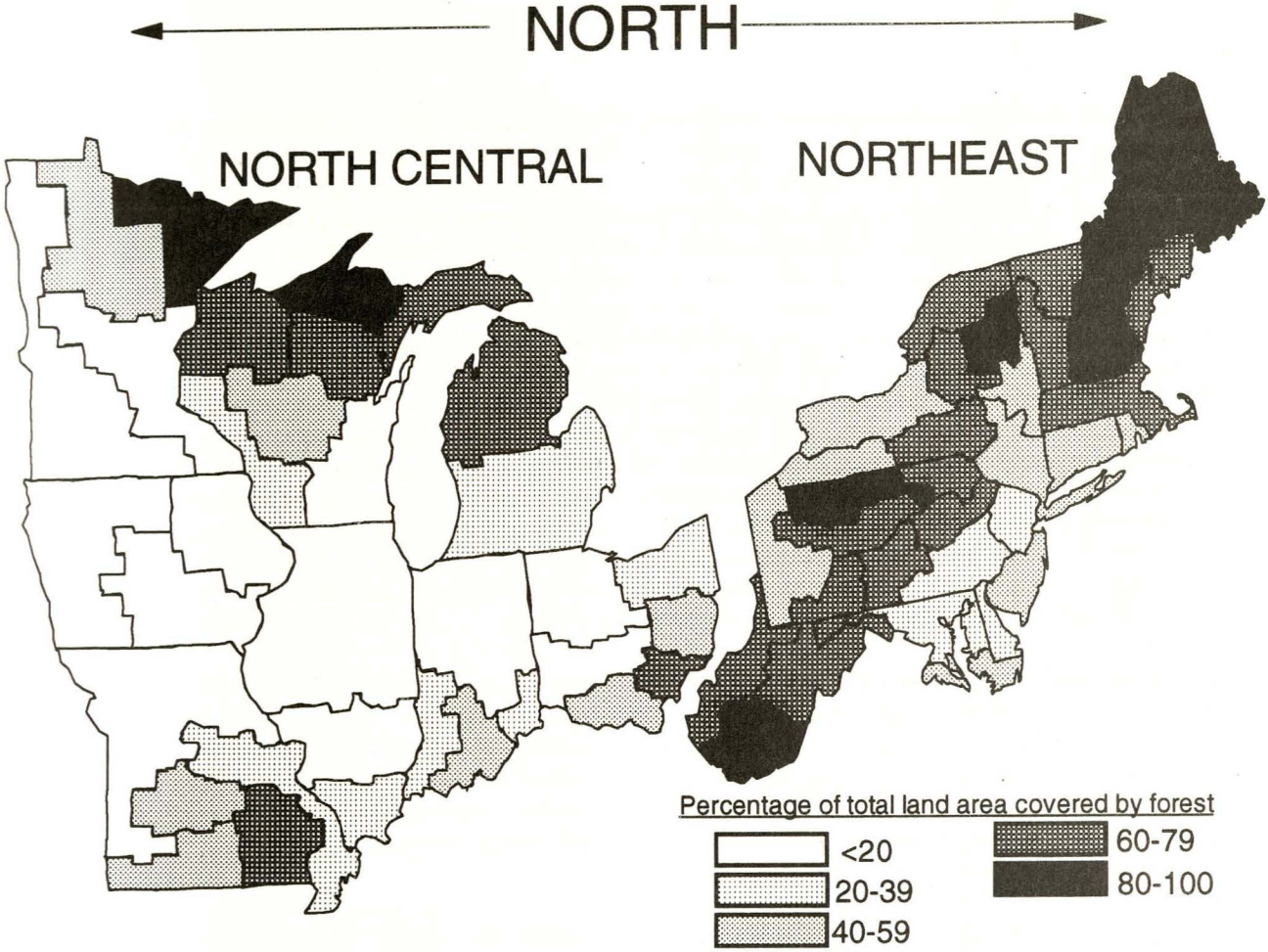


Figure 1. Northern Region and subregions used by the RPA Assessment and percentage of land area covered by forest, by FIA geographic unit, current inventory.

preserves or urban forests. The future area of unproductive forest is expected to remain relatively static over time as this category is more dependent on site capability than land use. Thus, future fluxes (both positive and negative) in the area of forest land are expected to occur in the timberland category. However, despite FIA classifications and definitions related to timberland, timber production often is not the primary reason for ownership for many timberland owners (Birch 1996). Because of this, it is important to determine the extent and character of today's timberland to improve future forest management decisions.

## FOREST-LAND AREA

The most recent estimates of forest-land area by FIA units reveal that there are 169.4 million acres of forest land in the 20 northern states—85.5 million in the 12 Northeastern states and 83.9 million in the 8 North Central states. These forests cover 67 percent of the land area in the Northeast and 29% of the land in the North Central states. Recent inventories have shown increases in forest land area since the previous inventory in most states across the Northern Region, the exceptions being the most recent inventories of Connecticut and Delaware, where forest land decreased—by 2 and 1%, respectively. In the Northeast, recent increases in forest have been a continuation of past trends. Whereas, in many of the North Central states forest area has been rebounding after declining between 1952 and 1977.

Recent changes in land-use (primarily agricultural) practices and policies have more than offset losses due to development for nonforest uses. Examples of such changes include the abandonment of cropland/pasture and subsequent reversion to forest; reduction in the number of farming operations and subsequent increase in average farm size, resulting in less intensive management of noncultivated areas and enabling the reestablishment of forest;

changes in livestock grazing management that include greater reliance on feedlots rather than fields and pastures; increased wildfire control; and government programs, such as the Conservation Reserve Program, which remove highly erodible lands from crop production and enable the land to revert to forest or grassland. For the entire Northern Region, forest-land area has increased between the most current and previous inventories by an average of 3.5%. This includes increases of 1.5% in the Northeast and 5.7% in the North Central states.

## TIMBERLAND AREA

Timberland represents 94% the total area of forest land in the Northern Region. Recent inventories in the Northeast have found a slowing in the amount of farm land being abandoned while the forest acreage reserved from cutting or classified as urban forest has increased steadily with each new inventory. These shifts out of timberland and into the noncommercial forest category, along with the continued loss of forests to development for nonforest uses, have reduced the amount of timberland in many Northeastern states (Table 1). Although difficult to predict, shifts of timberland into a reserved or noncommercial category may significantly influence the future timber resource in some Northeastern states. In densely populated states such as New Jersey and Connecticut significant amounts of forest land could be reclassified as urban forest by future inventories.

In the North Central states, the area of forest land and timberland has increased during the latest inventory period. For example, the 1993 inventory of Michigan reported an increase of more than a million acres in timberland. Most of this increase was due to changes in agricultural practices, primarily abandoned cropland reverting to forest land. Whether these regional trends will continue depends on a number of factors. Also significant in Michigan and Minnesota, a large amount of land classified by earlier inventories as nonproductive forest (not timberland) was reclassified as marginally productive timberland during the most current inventory.

Table 1. Current estimates of timberland area and changes in timberland area between the current and previous inventory, Northern Region.

| Northeastern states/ inventory period | Acres of timberland (Millions) | Percent change | North Central states/ inventory period | Acres of timberland (Millions) | Percent change |
|---|---|---|---|---|---|
| Connecticut 1972-85 | 1.8 | -1.6 | Illinois 1962-85 | 4.0 | +1.2 |
| Delaware 1972-86 | 0.4 | -2.1 | Indiana 1967-86 | 4.3 | +10.3 |
| Maine 82-95 | 16.9 | -1.1 | Iowa 1974-90 | 1.9 | +33.2 |
| Maryland 1976-96 | 2.4 | -3.9 | Michigan 1980-93 | 18.6 | +6.5 |
| Massachusetts 1972-85 | 2.9 | +4.7 | Minnesota 1977-90 | 14.7 | +8.2 |
| New Hampshire 1973-83 | 4.8 | +2.6 | Missouri 1972-89 | 13.4 | +8.2 |
| New Jersey 1974-87 | 1.9 | +0.4 | Ohio 1979-91 | 7.6 | +9.4 |
| New York 1980-93 | 15.4 | -0.3 | Wisconsin 1968-83 | 14.8 | +1.5 |
| Pennsylvania 1978-89 | 15.9 | -0.3 | **Total North Central** | **79.3** | **+6.8** |
| Rhode Island 1972-85 | 0.4 | -6.0 | | | |
| Vermont 1973-83 | 4.4 | -0.2 | **Northern Region** | **158.5** | **+3.6** |
| West Virginia 1975-89 | 11.9 | +3.8 | | | |
| **Total Northeastern** | **79.1** | **+0.3** | | | |

Timberland in the Northern Region is dominated by hardwood forest types that cover 81% of the regions timberland. Hardwood forest-type groups include the following: Oak/Pine, Oak/Hickory, American Elm/Green Ash, Oak/Gum/ Cypress, Northern Hardwoods (Maple/Beech/Birch), and Aspen/Birch. Current inventories show large increase in area for the Northern Hardwood group across the region.

## STAND STRUCTURE

Timberland can be classified according to the size of the trees growing on the land: sawtimber stands, poletimber stands, sapling/seedling-stands, and nonstocked areas (Figure 2). These size classes are based on commercial uses for forest products and also relate to the seral stage of the forest and habitat requirements of wildlife species. Sawtimber stands, which have the majority of their stocking in sawtimber-size trees, cover 52% of hardwood typed timberland in the Northeast Region and 49% of the hardwood-typed timberland in the North Central states. Sawtimber-size stands have increased in extent primarily because poletimber stands have grown into the sawtimber-size class. Sawtimber stands are most attractive to forest industry because of the high volume and value associated with potentially higher quality trees (size is a major factor in determining quality for hardwood trees). Stands dominated by large trees are aesthetically pleasing—people enjoy hiking and camping in such stands because they are easier to move around in and are perceived to be more attractive. Sawtimber stands also benefit a variety of wildlife species that require stands with large trees for at least part of their habitat or life cycle.

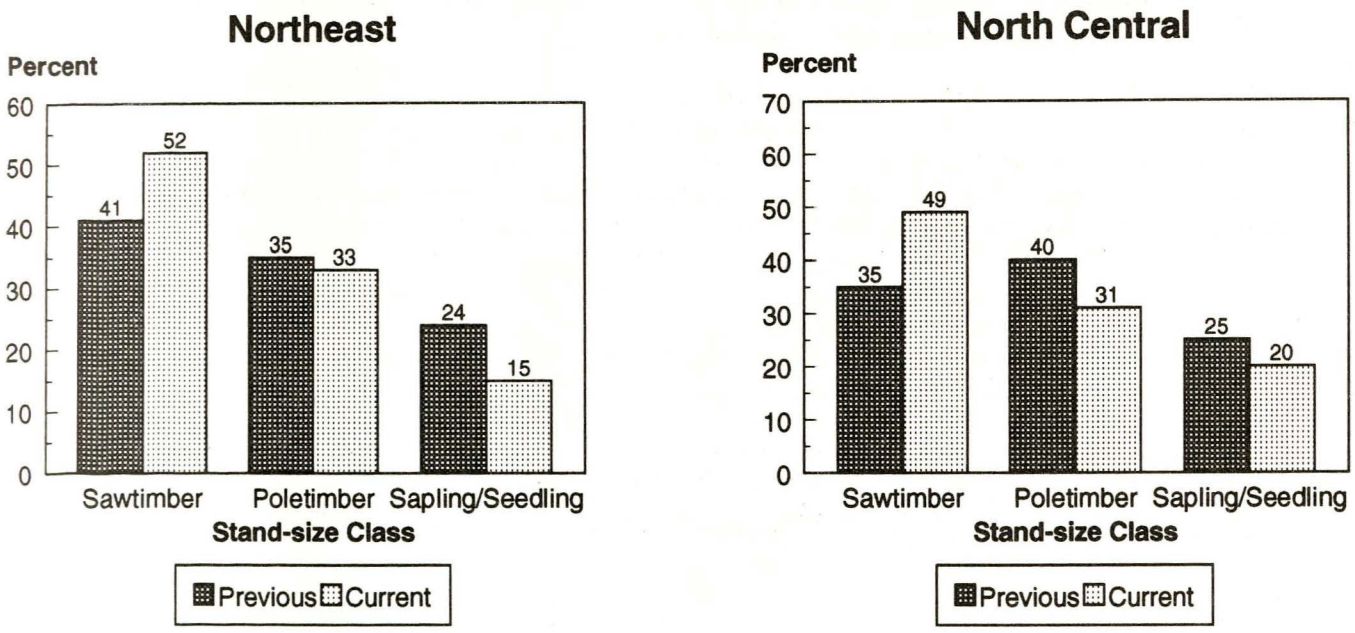


Figure 2. Change in stand-size class between previous and current inventories, on timberland classified in a hardwood forest type, Northeast and North Central subregions.

Poletimber-size stands cover about a third of the timberland in the Northeastern and North Central states. Poletimber stands have the potential for rapid increases in value as they grow into sawtimber stands. Silvicultural treatments in these stands focus primarily on enhancing future value. Historically, crop tree release, thinnings, and improvement treatments in hardwoods have not been common in the Northern Region. Poletimber-size stands are associated with fewer wildlife species than either regenerating stands (sapling-seedling class) or sawtimber stands (DeGraaf and others 1992). Trees in these stands often lack the maturity to produce hard mast or to provide opportunities for species that use the boles of trees. Also, the dense closed overstory can inhibit the growth of herbaceous plants and shrubs that provide food and cover for wildlife.

Sapling-seedling stands and nonstocked areas usually result from clearcutting, salvage cutting, or agricultural land that has been abandoned. Such stands decreased in both the Northeastern and North Central states, relative changes were -36% and -19%, respectively. Across the Region, growth of these stands into the poletimber class was not offset by regeneration harvests or farmland reverting to forest land. In the North Central states the reversion of farm

land offset growth into larger-size stands to a greater extent than in the Northeastern states. The nature and extent of harvesting activities across the Northern Region are not well documented, but FIA plot information reveals that contemporary harvesting activities are not reducing many stands to these early successional stages. In an analysis of FIA plot data in Pennsylvania for 1978-1989 (Gansner and others 1993), they found that on 90% of timberland, removals amounted to less than 40% of the original basal area. As the abandonment of farm land slows and clearcutting becomes even less favored as a harvesting method, the area in sapling-seedling stands will continue to decrease.

The reduction in sapling-seedling stands represents a decrease in the area of early successional stage forest. Because some wildlife species depend on these stages as nesting or feeding sites, it is desirable to have a portion of the forest in sapling/seedling-size stands. The decline of early successional forests across the Northeast Region is of concern to many wildlife biologists. For example, Brooks (1993) stated that in eastern North America, declines are occurring in several bird species that breed in young forests. He added that "although the loss of young and old-field, or shrubby, thicket-type forests cannot be solely blamed for the declines, loss of summer-breeding habitats is certainly a partial cause of these trends." Across a large region it is also desirable to have a mix of stand sizes. A forest with a balanced distribution of size classes provides diverse habitats and an even flow of forest products, and also may be more resistant to insects and diseases than less balanced forests.

## CHANGES IN TIMBER VALUE

The value of the timber resource depends on the total volume of wood available and the character of the trees. Key characteristics in valuing the hardwood resource are species, tree size, and quality.

The fundamental measure of timber volume used by FIA is the net cubic-foot volume of wood in the main stem of well-formed sound trees that are 5 inches or more in diameter—growing-stock volume. As hardwoods are the predominate species group in the North, they represent three-fourths of the growing-stock volume. The volume of hardwoods has increased steadily in both the Northeastern and North Central States: the current estimated volume of hardwood growing-stock in both areas is more than twice what it was in 1952 (Figure 3).

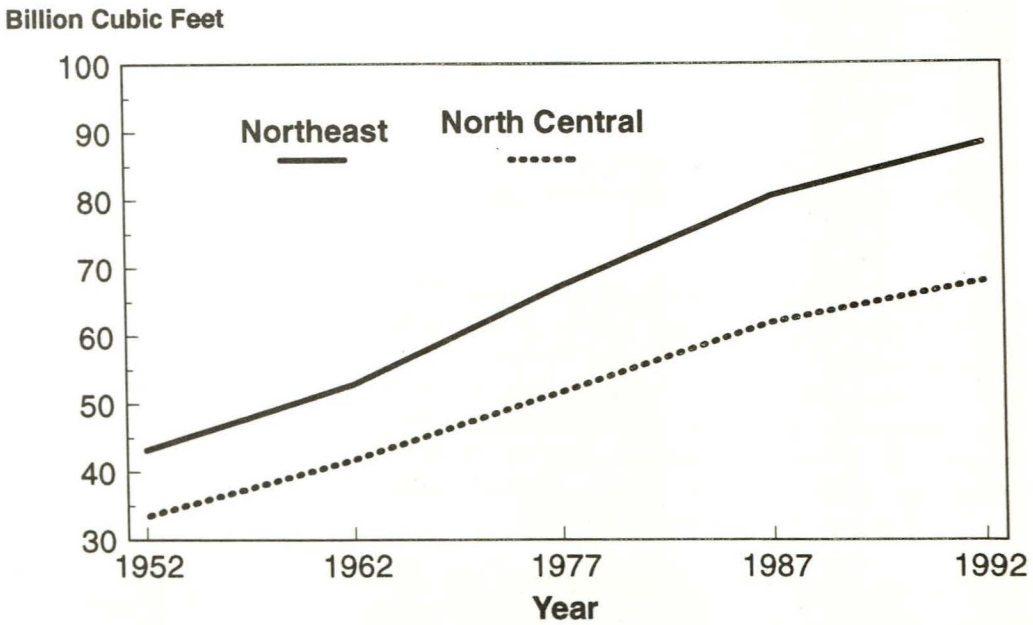


Figure 3. Hardwood growing-stock volume on timberland, Northeastern and North Central states.

Putting volume on a per-acre basis allows comparison of areas with different amounts of timberland (Figure 4). The highest hardwood volumes per acre are in the Appalachians, particularly on the Allegheny Plateau in northwestern Pennsylvania. High per acre volumes are also found near some of the more heavily populated areas of the Northeast. This implies that trees in these areas are valued for their nontimber benefits and that access to them by harvesters is more difficult than in more rural areas. This clash in forest values can be seen in the rise in the number of local cutting ordinances enacted by many communities. Farm woodlots across northern Ohio and southern Michigan also have large amounts of hardwoods, though these acreages are small. Lower hardwood volumes are found where forests are primarily conifers, such as in Maine and the northern areas of Michigan, Wisconsin, and Minnesota, and also where trees give way to shrubs and grasses due to climatic factors, for example, prairie regions in western Minnesota, Iowa, and Missouri.

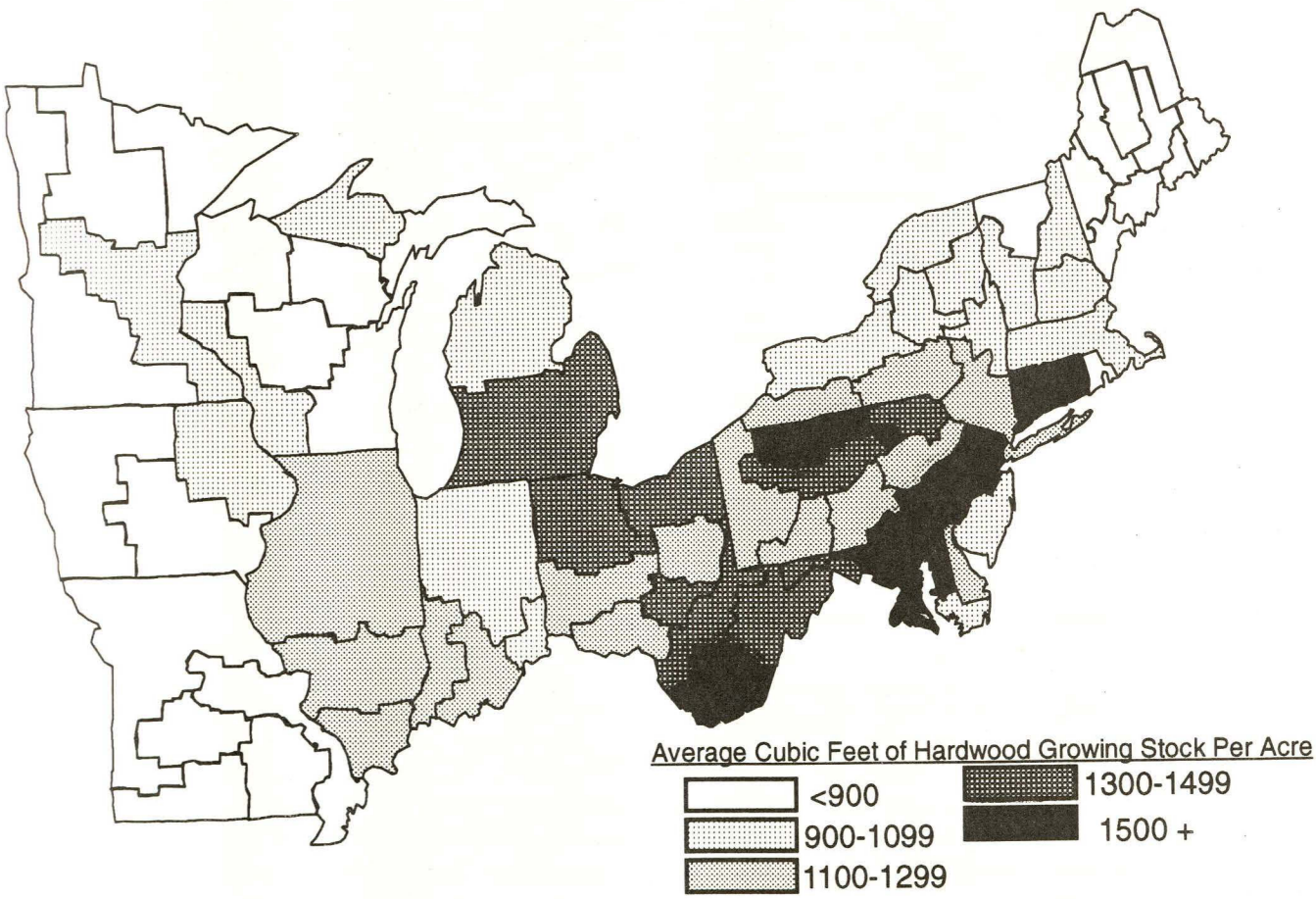


Figure 4. Average hardwood growing-stock volume per acre of timberland, by FIA geographic unit, Northern Region, current inventory.

Figure 5 shows the volume for all hardwood species by diameter class using FIA data compiled for the 1992 Resource Planning Act Assessment (Powell and others 1993). Since 1952, hardwood volume has increased steadily with each new assessment period. The increase in volume has not been distributed evenly across diameter classes. Volume increases were concentrated on sawtimber-size trees. Volume decreased in the 6-inch class for the two most recent assessments. In general, the larger the diameter, the larger the percentage increase in volume. This is characteristic of a maturing resource. The primary reason for the decrease in volume in the 6-inch diameter class was most likely due to a lack of recruitment into this small-diameter class, which probably has resulted from increased competition from larger size trees. Recent inventories show the portion of volume growth coming from

accretion on remeasured trees has increased and the portion of volume growth coming from the ingrowth of new trees into the lower diameter classes has decreased.

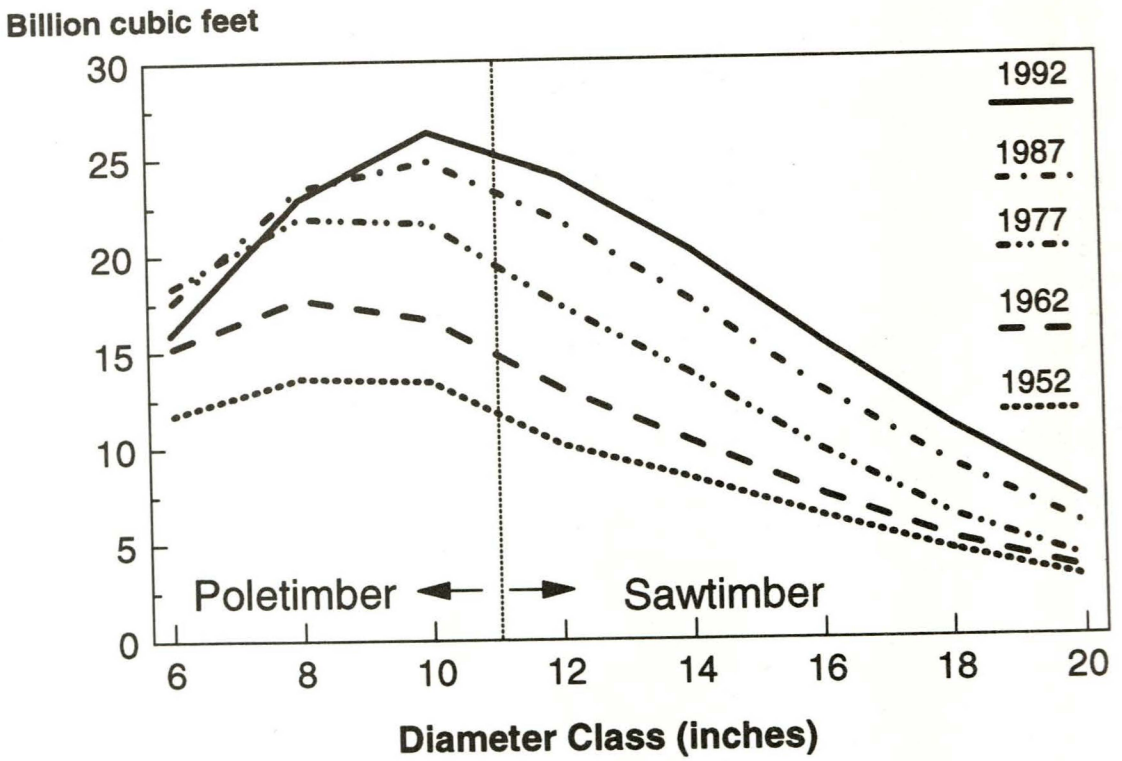


Figure 5. Hardwood growing-stock volume by diameter class, Northern Region.

From 1987 to 1992, 94% of the increase in growing-stock volume was in trees large enough to be sawn into lumber. Because so much of the growth is now occurring on large trees, the sawtimber portion of the resource, expressed in board feet (International 1/4-inch rule), has soared in many states (Table 2). This has added tremendous value to the hardwood resource. To a large extent, the quality of small diameter hardwoods is determined by diameter alone. The large numbers of trees growing into the sawtimber-size class and also small-sawtimber-size trees moving into the larger-sawtimber sizes has overwhelmed many of the negative effects of high-gradeing by the selective logging of large trees over the Region.

More than 100 species of hardwoods grow in the Northern Region. It is not uncommon to find many hardwood species growing on the same acre or in the same stand. However, the 10 leading species and species groups in terms of volume account for two-thirds of the total hardwood volume (Figure 6). The volume for all hardwood species increased by 32% over the last inventory cycle, but the increase was not distributed evenly across all species. The different rates of change for individual species indicate that the species composition of the forest is changing.

Table 2. Current hardwood board foot volume and percentage change from previous inventory

| Northeastern states | Current volume (Billions of board feet)* | Percent change | North Central states | Current volume | Percent change (Billions of board feet)* |
|---|---|---|---|---|---|
| Connecticut | 6.3 | +48 | Illinois | 17.2 | +57 |
| Delaware | 1.3 | +31 | Indiana | 18.6 | +51 |
| Maine | 17.1 | +9 | Iowa | 5.7 | +53 |
| Maryland | 11.1 | +22 | Michigan | 48.3 | +58 |
| Massachusetts | 5.8 | +66 | Minnesota | 22.7 | +43 |
| New Hampshire | 8.5 | +50 | Missouri | 23.1 | +49 |
| New Jersey | 4.4 | +40 | Ohio | 29.3 | +66 |
| New York | 43.6 | +46 | Wisconsin | 25.1 | +60 |
| Pennsylvania | 59.5 | +37 | **Total North Central** | **189.1** | **+55** |
| Rhode Island | 0.7 | +57 | | | |
| Vermont | 8.9 | +29 | **Northern Region** | **410.3** | **+47** |
| **West Virginia** | **54.0** | **+61** | | | |
| **Total Northeastern** | **221.2** | **+41** | | | |

*International 1/4-inch rule

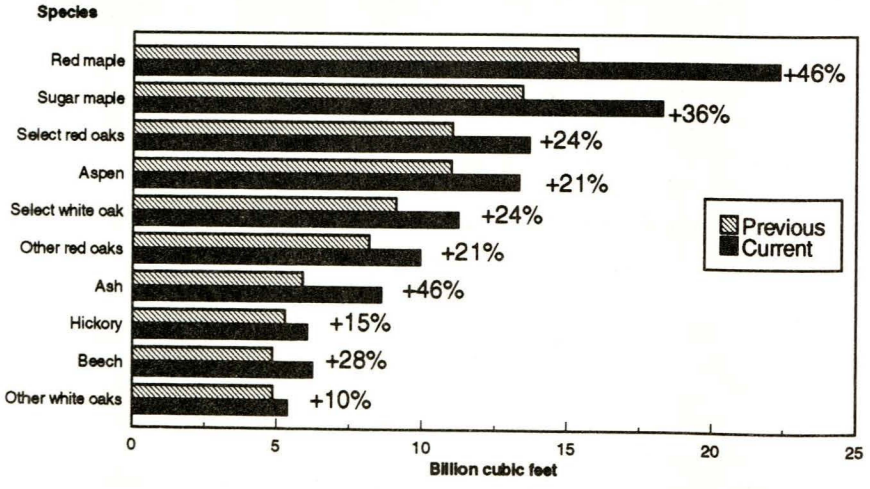


Figure 6. Change in growing-stock volume between previous and current inventories, for selected species and species groups in the Northern Region.

Red maple (*Acer rubrum* L.) led the region in volume at both the most current and previous inventory periods, accounting for 14% of the total volume of all species. Red maple is distributed widely, is an aggressive competitor, and is not heavily cut. It had the largest percentage increase (46) as well as the largest increase in magnitude (7.0 billion cubic feet). Sugar maple (*Acer saccharum* Marsh.) also had larger increases in volume both in percent (36) and magnitude (4.9 billion cubic feet). Together, maples increased as a proportion of the total resource—representing 24% of the previous inventory versus 26% of the current. The four groups of oak species had smaller increases. Across the region, high mortality and reductions in growth have had negative impacts on the oaks. Some reasons for reductions in the growth of oaks are defoliation by the gypsy moth caterpillar, high cutting rates, and low ingrowth due to poor regeneration. Few stands in the Alleghenies (Allegheny Plateau and Allegheny Mountain sections of the mid-Atlantic states) contain sufficient numbers of oak seedlings to adequately regenerate oak after cutting (Marquis and others 1992)

The use of the timber resource for timber products is determined largely by the tree quality. The best trees are used for the manufacture of furniture, cabinets, and other millwork, and lower quality trees are used for pallets, pulpwood, and particle board. Quality is indicated by the grade assigned to each tree on inventory sample plots. Tree grade is based on the number of clear surfaces, amount of cull, and diameter of the butt log of the tree. Tree Grade 1 yields the most high-grade lumber and Tie/Timber Grades the least. The quality of trees varies with species. Figure 7 compares tree grades for five major species in Ohio. Only trees greater than 15 inches in diameter are used in order to reduce the influence that size alone has on the quality of small sawtimber-size trees. Ohio data are used here as an example of trends taking place across the Northern Region. Ohio is similar to the Northern Region as a whole in that the portion of total volume of sugar and red maple has increased at the expense of more desirable species such as red and white oaks.

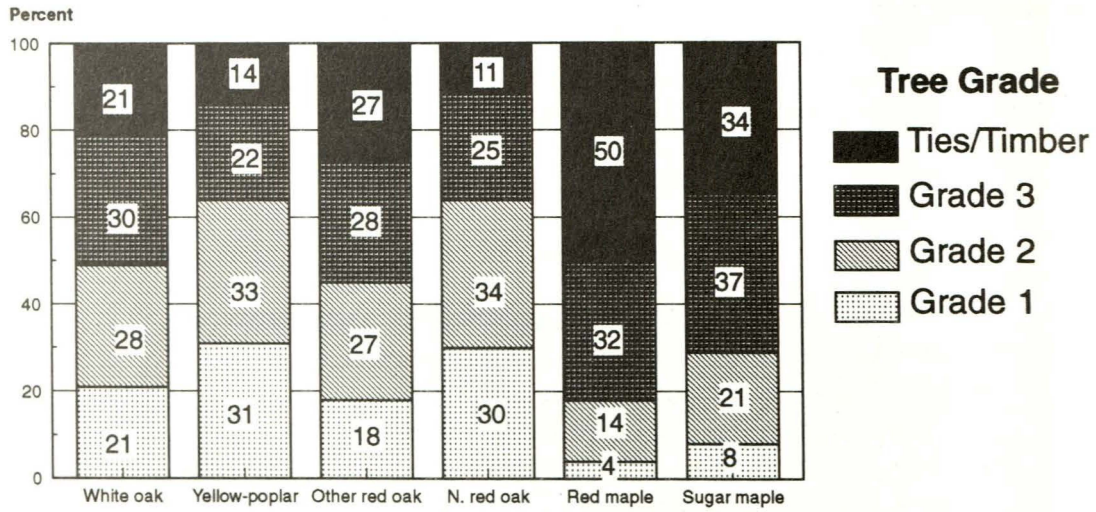


Figure 7. Percentage of sawtimber volume in each tree grade for trees 15 inches and larger in diameter, for selected species, Ohio, 1991.

Sixty-four percent of the northern red oak (*Quercus rubra*) volume was in Tree Grade 2 or better trees, whereas only 18% of the red maple volume is in the better grades. These differences in grade between species have important implications for future supplies of high-quality lumber. Because future forests will contain significantly more red maple sawtimber volume and a lower proportion of volume in oak species, the yield of high-quality lumber from these forest will not increase proportionally with increases in total volume due to differences in species characteristics.

Shade-tolerant species tend to retain their lower limbs longer, thereby reducing the yield of high-quality lumber; whereas shade-intolerant species naturally prune their limbs at an earlier age, which results in fewer and smaller knots. In general, the more shade-tolerant species, such as red maple do not grade out as well as intolerant species such as red oak. The utilization of low-quality hardwood trees has caused persistent problems for forest managers, and these problems will probably continue as long as red maple continues to increase in volume. Although great strides have been made in utilization in recent years, there remain tremendous challenges and opportunities.

## GROWTH AND REMOVALS

A look at the relationship between net growth and removals of growing-stock volume helps in understanding the sizable increase in hardwood growing-stock volume (Figure 8). Average annual net growth includes accretion (growth on the initial inventory), ingrowth (volume of trees that grew large enough to measure), mortality, and changes in cull volume. Red maple surpassed all other species in net growth, accounting for 17% of the total volume of net growth for the region. Net growth for the four oak species groups combined accounted for 22% of total net growth. Because of the higher removals rate for the oaks, the net change (net growth minus removals) for red maple was greater than for the oaks combined (551 million cubic feet for red maple versus 336 million cubic feet for all oaks combined).

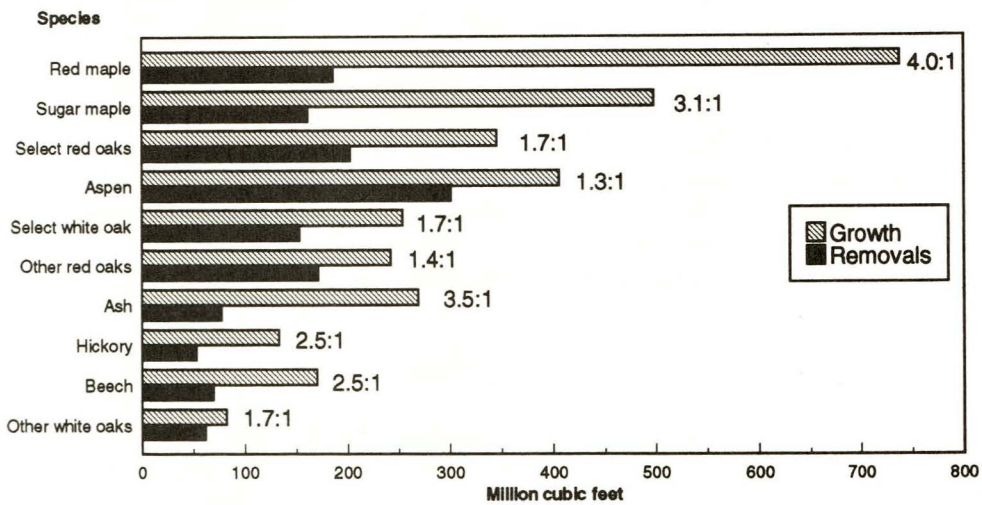


Figure 8. Average annual growth and removals of growing stock in the Northern Region, 1979-1991.

Dividing average net annual growth by average annual removals gives a ratio of growth to removals. Across the Northern Region, this ratio for hardwoods is 2.2:1. Ratios vary considerably between species. Red maple's high growth and relatively low removals result in a ratio of growth-to-removals of 4.0:1. This is almost twice the average for all hardwood species, and much higher than for the oaks (1.7:1 for both select red oaks and select white oaks). Comparing individual species to the average for all hardwood species (2.2:1) gives an indication of which species are increasing in importance, and which are decreasing. Red maple clearly will play a larger role in forests of the Northern Region, especially if the oaks continue to fade in importance.

Ratios of net growth to removals also vary considerably among states (Table 3). Although more hardwood volume is grown each year than is being cut in all states, the ratios are lower for the North Central than the Northeastern states. This reflects more intensive use of the resource in the North Central states.

Table 3. Ratio of growing stock growth to removals by state

| Northeastern states | Ratio of growth/ removals | North Central states | Ratio of growth/ removals |
|---|---|---|---|
| Connecticut | 2.5:1 | Illinois | 1.4:1 |
| Delaware | 2.0:1 | Indiana | 1.6:1 |
| Maine | 1.6:1 | Iowa | 1.8:1 |
| Maryland | 1.9:1 | Michigan | 2.7:1 |
| Massachusetts | 4.5:1 | Minnesota | 1.7:1 |
| New Hampshire | 3.8:1 | Missouri | 2.2:1 |
| New Jersey | 4.9:1 | Ohio | 2.3:1 |
| New York | 2.9:1 | Wisconsin | 1.6:1 |
| Pennsylvania | 2.0:1 | **Total North Central** | **2.0:1** |
| Rhode Island | 2.9:1 | | |
| Vermont | 4.1:1 | **Northern Region** | **2.2:1** |
| West Virginia | 3.7:1 | | |
| **Total Northeastern** | **2.6:1** | | |

## SUMMARY

Forests in the Northern region are maturing, as evidenced by increasing area of sawtimber stands, increases in the number of larger trees, higher volumes per acre, and changes in species composition toward more shade-tolerant species such as red and sugar maple. Although it is natural for tolerant species to increase in a maturing forest, higher cutting rates for the intolerant oaks probably have accelerated this process. Large increases in sawtimber volume have brought increases in the timber value of the resource, though changing species composition will slow such increases. Because red maple has increased its share of the total resource at the expense of the more desirable oaks, increases in volume have not been accompanied by proportional increases in timber value.

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# PRIVATE FOREST OWNERS OF THE CENTRAL HARDWOOD FOREST

Thomas W. Birch[1]

Abstract: A recently completed survey of woodland owners provides insight into the owners of private forest lands in the Central Hardwood Region. There is increasing parcelization of forested lands and an increase in the numbers of nonindustrial private forest-land owners. Over half of the private owners have harvested timber from their holdings at some time in the past, they control three-quarters of the private forest. Owners have a positive attitude toward timber cutting at a time when there is greater demand for products from the forest. Because private forest owners have virtually total authority over their lands, it is essential that we monitor this resource and maintain good stewardship to ensure continued productivity for future generations.

## INTRODUCTION

The 14 state Central Hardwood Region (Figure 1) has 136.0 million acres of forest land (36% of the total land area). Approximately 100 million acres of private forest are controlled by an estimated 3,179,900 private ownerships. While these owners hold forest land for a variety of reasons, most have positive views on timber harvesting. Resource professionals and policymakers are determined not to repeat past mistakes: they want to ensure that renewed interest in timber harvesting is based on principles of good stewardship. The forest-based community needs to keep close watch on forest management activities in these states.

A significant development in the management of forests in the Central Hardwood Region has been an increase in the amount of smaller acreage forest ownerships. Since 1978, forest land among ownerships of fewer than 50 acres increased from 12 to 30 million acres; among ownerships of 50 to 99 acres, forest area increased from 12 to more than 18 million acres. Meanwhile forest land among ownerships of 100 to 1,000 acres decreased from 46 to 35 million acres since 1978. These are some of the findings in "Private forest-land owners of the United States, 1994" (Birch 1996). This study was undertaken as a joint effort between the research and state and private branches of the USDA Forest Service, and the National Association of State Foresters. It is the second effort at national coverage since 1978 (Birch and others 1982).

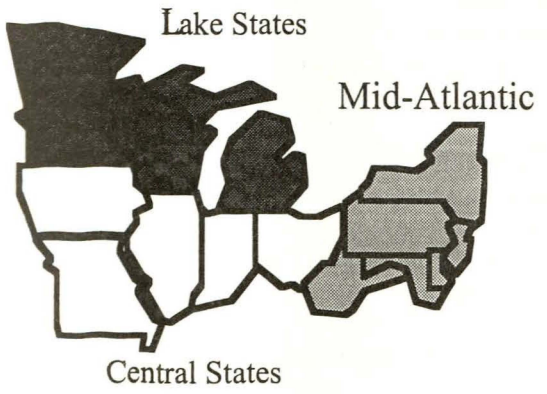


Figure 1. States included in the Central Hardwood Region.

[1]Resource Analyst, USDA Forest Service, Northeastern Forest Experiment Station, Radnor, PA 19086.

The 1994 study has some new features: American Indian and Native Tribal Lands formerly classified as federal lands are now recognized as private lands, and the motivations and management intentions of forest-land owners, including timber-harvesting behavior, are explored in detail. The new study also provides reliable results at the state level.

## METHODS

The 1994 study was conducted by the USDA Forest Service's Forest Inventory and Analysis (FIA) units in cooperation with the National Association of State Foresters and the USDA Natural Resources Conservation Service, in support of the Forest Stewardship Program of the USDA Forest Service, State and Private Forestry. The questionnaire used in this study was mailed to 13,333 owners of 14,619 private forested sample plots. Sample plot locations were from National Resources Inventory (NRI) plots or FIA plots. Responses from 6,224 ownerships that hold 7,125 of these sample plots are within the Central Hardwood Region.

The questionnaire was derived from earlier ownership studies and has been revised over 20 years for the Northeastern and North Central states. The probability that a forest-land owner will be sampled depends on the rate of sampling and the amount of acreage of forest land owned. Each state and often areas within states have different rates of sampling and response rates differ. This affects the probability that an owner is included in the final tabulations. The total acreage of forest land for a state or sampled area is obtained from the forest inventory. Since the sampling scheme is essentially the one used for the forest inventory, there is a low probability that owners of small parcels of forest land will be included. To estimate the total number of persons or ownership units (Birch 1996), it is necessary to weight the number of owners based on their probability of selection. The summation estimate equals the estimated number of ownership units in the state or region.

Periodic inventories of forest resources by FIA identify ownership of inventory sample plots in broad categories such as forest industry, farmer, corporate, and "other" private owners. In turn, estimates of the total number of acres in each category by geographic area are provided. FIA crews do not interview individual landowners on which inventory plots are located to determine how many acres of forest land each owns. Hence, only a land-ownership study can provide information such as number of owners, their distribution by ownership size class, and their reasons for owning forest land and plans for future use.

## RESULTS

There are currently 9.9 million private forest-land ownerships in the United States, an increase of 2.1 million (28%) from the 7.8 million owners estimated in the 1978 study. In the Central Hardwood Region, ownerships increased from 2,620,400 in 1978 to 3,181,000 in 1994. Area of private forest increased from 86 million acres in 1978 to 100 million acres in 1994. Private forest ownerships are well distributed throughout the 14 states. Pennsylvania has 514,000 owners while Delaware has 17,000 (Table 1).

Individual ownerships are by far the most common form of forest-land ownership across the Central Hardwood states (and the nation). As used here individual ownership includes not only land held by a single person but also joint ownership, such as husband and wife or relatives, as distinct from corporations, partnerships, estates, clubs, and trusts. While 94% of the ownerships are classified as individual owners, they control 76% of the private forest land in the Central Hardwood Region (Figure 2). By contrast, corporations account for 2% of the ownerships, but collectively own 15% of the private forest. Partnerships, clubs, etc. make up the remaining 4% of the private ownerships controlling 9% of the private forest.

Table 1. Owners and acres of forest land by state and region, 1994

| State | Owners | | Acres | |
|---|---|---|---|---|
| | Thousand | Percent | Million | Percent |
| Delaware | 17 | 0.5 | 0.35 | 0.3 |
| Maryland | 126 | 4.0 | 2.27 | 2.3 |
| New Jersey | 89 | 2.8 | 1.40 | 1.4 |
| New York | 490 | 15.4 | 14.37 | 14.4 |
| Pennsylvania | 514 | 16.2 | 12.51 | 12.5 |
| West Virginia | 260 | 8.2 | 10.75 | 10.8 |
| Mid-Atlantic | 1,497 | 47.1 | 41.65 | 41.7 |
| Illinois | 114 | 3.6 | 3.64 | 3.6 |
| Indiana | 151 | 4.7 | 3.77 | 3.8 |
| Iowa | 56 | 1.8 | 1.80 | 1.8 |
| Missouri | 307 | 9.7 | 11.64 | 11.6 |
| Ohio | 329 | 10.3 | 7.19 | 7.2 |
| Central states | 957 | 30.1 | 27.94 | 28.0 |
| Michigan | 333 | 10.5 | 12.05 | 12.1 |
| Minnesota | 147 | 4.6 | 7.33 | 7.3 |
| Wisconsin | 246 | 7.7 | 10.83 | 10.8 |
| Lake States | 726 | 22.8 | 30.21 | 30.2 |
| Total | 3,181 | 100 | 99.90 | 100 |

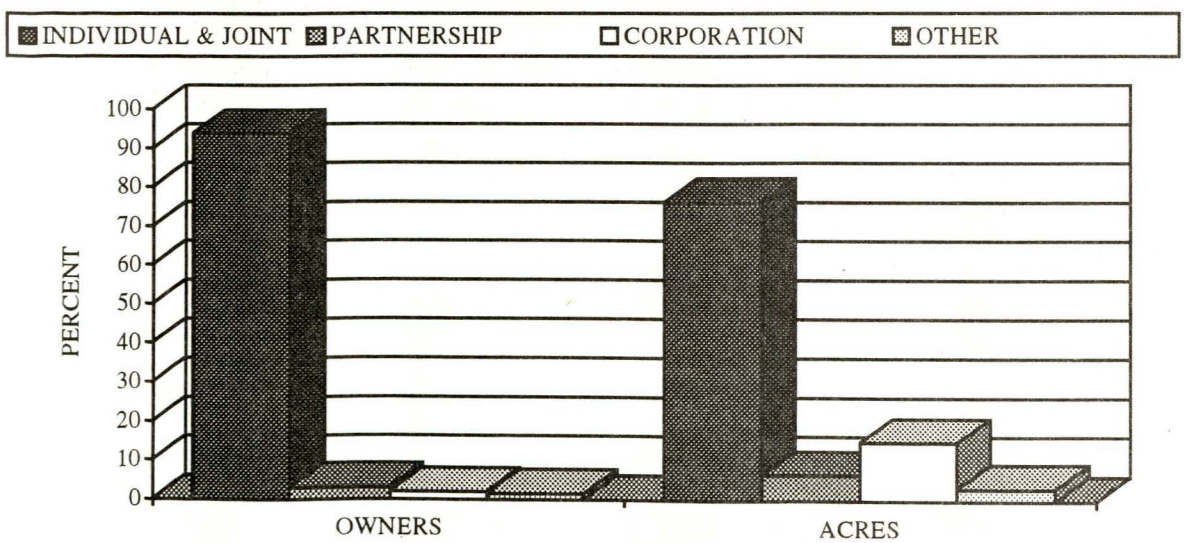


Figure 2. Distribution of private ownerships and private forest land, by ownership type, Central Hardwood Region, 1994.

Parcelization, the breaking up of larger forested tracts into smaller parcels, is occurring across the Central Hardwood Region. Forest land in ownerships smaller than 10 acres increased by 1.7 million acres between 1978 and 1994 (Table 2, Figure 3). More than 16 million acres were added to the 10- to 49-acre size class, a near tripling of the area in this class. Forest land in the 50- to 99-acre class increased by 6.3 million acres during the interval. Much of the additional acreage in small ownerships was from forest land held by owners of 100 to 499 and 500 to 999 acres. Forest land in these size classes decreased by 9.26 million acres. Among ownerships with more than 1,000 acres each, forest land increased by nearly 620,000 acres.

Table 2. Owners (thousands) and acres (millions) of private forest land, by size class of ownership, Central Hardwood Region, 1978 and 1994

| Size class (acres) | Owners* | | Acres* | |
|---|---|---|---|---|
| | 1978 | 1994 | 1978 | 1994 |
| 1-9 | 1,901.5 | 1,545.8 | 3.58 | 5.30 |
| 10-49 | 333.4 | 1,151.2 | 8.60 | 24.91 |
| 50-99 | 160.4 | 278.5 | 11.81 | 18.12 |
| 100-499 | 210.8 | 193.2 | 39.15 | 30.73 |
| 500-999 | 10.5 | 7.7 | 7.02 | 4.58 |
| 1,000+ | 3.7 | 3.4 | 15.69 | 16.31 |
| Total | 2,620.4 | 3,179.9 | 85.95 | 99.94 |

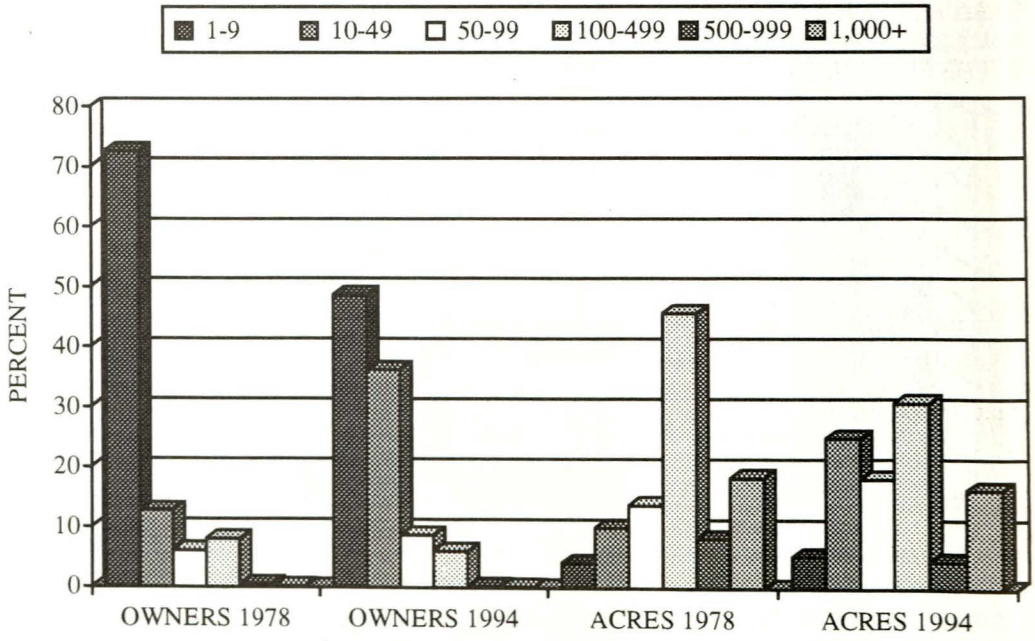


Figure 3. Distribution of private forest ownerships, by size class of ownership, Central Hardwood Region, 1978 and 1994.

Nationwide, the "new" (having first acquired forest land since 1978) individual private owner is younger and better educated, and has a higher income than the previous owner. Comparing occupation data from 1994 and 1978 (Birch 1996, Birch and others 1982) shows the changing profile of owners in the Central Hardwood Region. The new study (Birch 1996) shows an increase in the proportion of farmers, retirees, and other owners, including service workers (Figure 4). The proportion of acreage held by individuals increased among retired, white collar, and other owners. There were major decreases in area held by farmers and blue collar owners.

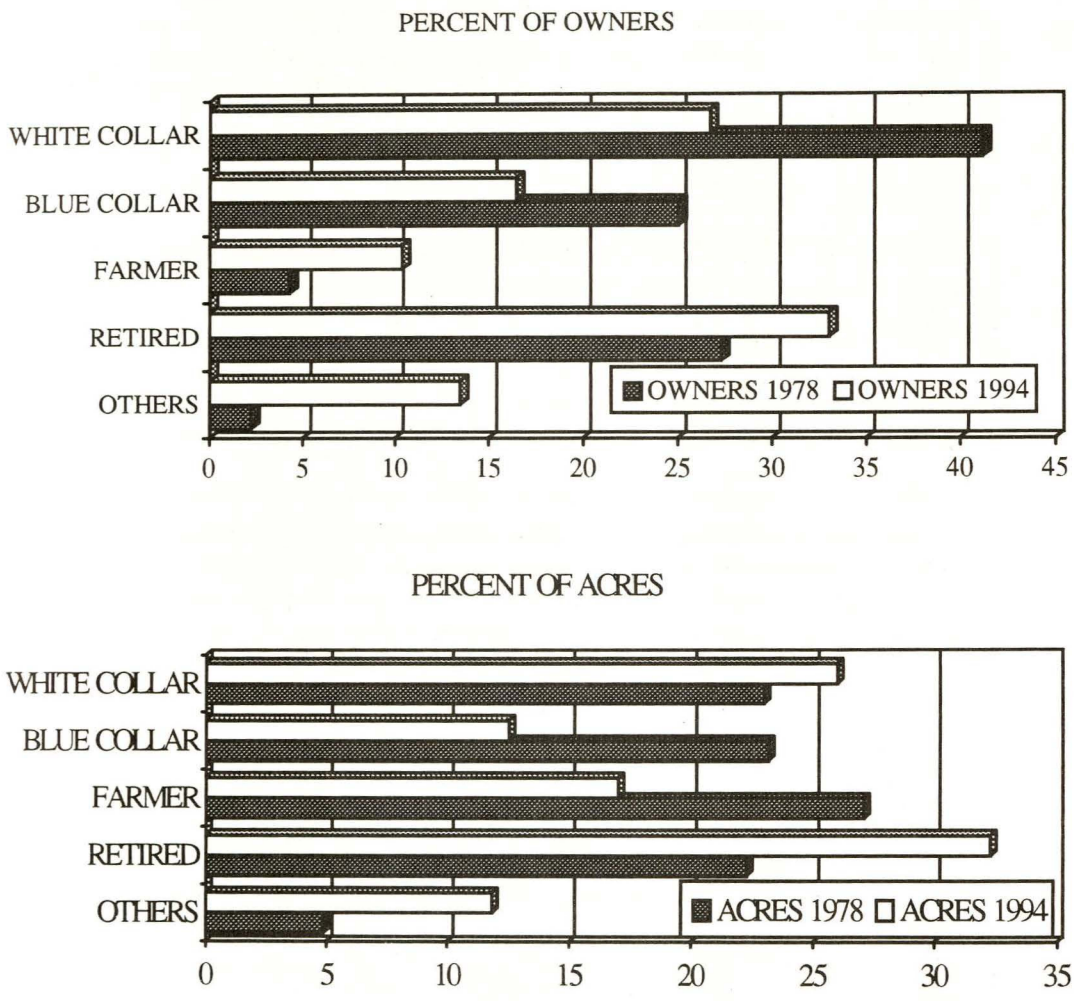


Figure 4. Distribution of individual ownerships, by owner occupation, Central Hardwood Region, 1978 and 1994.

It is estimated that 32% of all individual owners of forest land are over 65 years old. They own 38% of the private forest land in the Central Hardwood Region (Figure 5). The high proportion of older owners has serious implications for land tenure in the future. Much of the forest land controlled by older owners will change hands in the next decade. Individual owners under 45 years of age hold only 15% of the forest land owned by individuals. Owners 45 to 64 years of age control 43% of the forest land held by individuals.

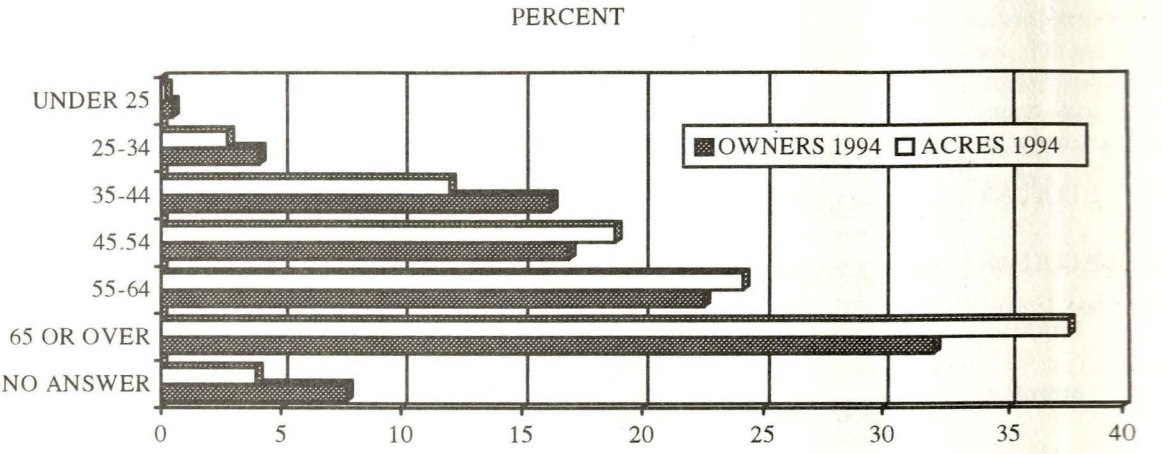


Figure 5. Distribution of individual ownerships, by owner age, Central Hardwood Region, 1994.

The yearly transition to many new forest owners makes it difficult to communicate information about management and new programs such as stewardship. The use of mass communication, especially the electronic media, is essential to reach these owners (Birch and Pywell 1986). Approximately 30% of the current owners acquired forest land for the first time since 1978, they control 23% of the private forest land (Figure 6). The largest decrease in numbers of owners was among those who first acquired forest land prior to 1950. The largest decrease in acreage owned was among those who had first acquired forest land between 1970 and 1977.

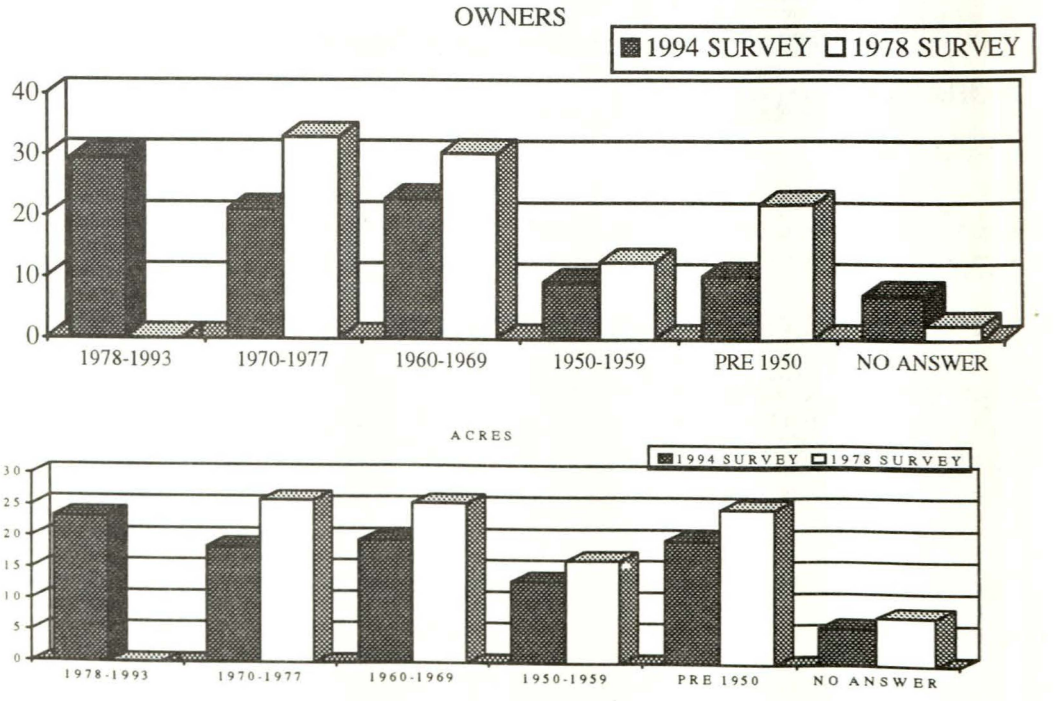


Figure 6. Distribution of private ownerships, by year owner first acquired forest land, Central Hardwood Region, 1978 and 1994.

Owner Objectives

Forests produce many benefits for their owners, so it is not surprising that we found diverse reasons for owning forest land. Some potential benefits can be produced with little or no effect on others, while other benefits even increase when another benefit is produced.

Nearly 40% of the private forest-land owners stated that their primary reason for owning forest land is that it is "part of the farm" or "residence" (Figure 7). In general, these ownerships hold smaller than average tracts. Another 11% of the owners stated that farm or domestic use is the most important reason for owning forest land. Many of these owners consider their woodland as a source of fenceposts, fuelwood, and similar products.

Recreation and esthetic enjoyment is the primary reason why 30% of the owners hold forest land. The area controlled by people with these objectives represents 29% of the private forest land.

Land investment often is thought of as a hedge against inflation. Private landowners who list investment as their primary reason for owning forest land account for 6% of the owners and 8% of the private forest.

Only 1% of the private forest-land owners hold their land primarily for timber production, but these owners control 11% of the private forest land. Some owners own forest land for the minerals under the surface, many of these owners are included under the other category. Also included under the other category are the lands owned by Indian Tribes as part of their cultural heritage.

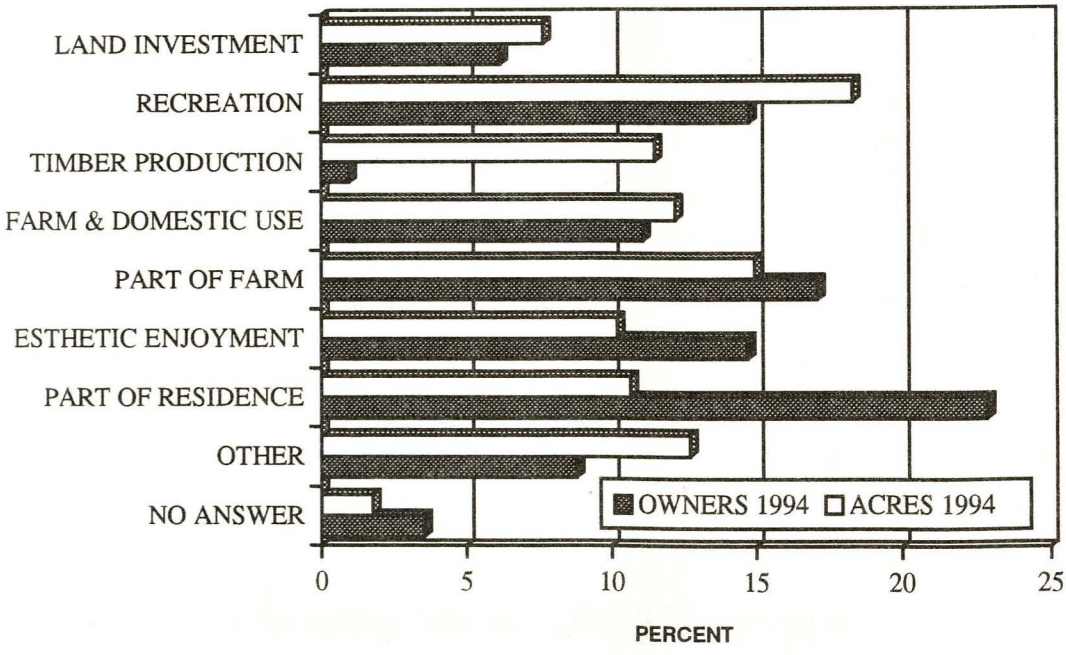


Figure 7. Distribution of private ownerships, by primary reason for owning forest land, Central Hardwood Region, 1994.

Timber Harvesting

Nearly one-half (48%) of the private owners in the Central Hardwood Region have harvested timber. These owners control 71% of the private acreage. By contrast, the 52% who have not harvested own only 29% of the forest land.

Those who stated that they intend to harvest in the next 10 years represent an estimated 36% of the private forest-land owners and 56% of the forest land. Conversely, the 29% who said they never intend to harvest own only 14% of the private acreage (Figure 8). Owners with indefinite harvest plans control 23% of the private forest land.

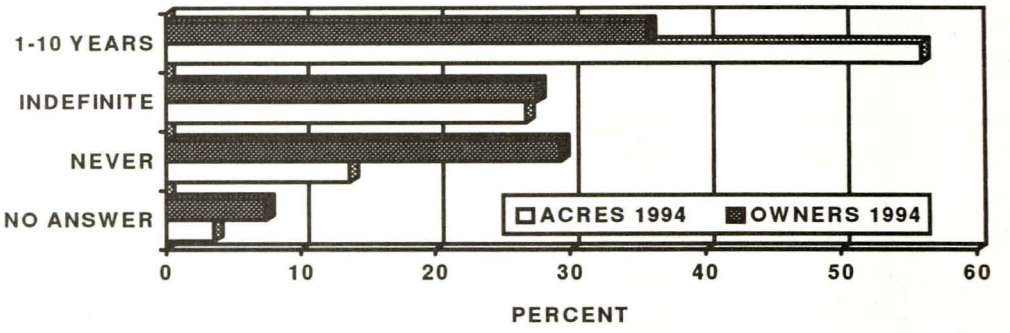


Figure 8. Distribution of private ownerships, by harvest intentions of private ownerships and acres of forest land owned, Central Hardwood Region, 1994.

Management Planning

A final measure of management activity is provided by the number of owners and acres of private forest land covered by written management plans. Approximately 5% of the private ownerships with 22% of the private forest land have some form of written management plan (Figure 9). Forest industry owns 25% of the forest land owned by those with a written plan. The other 75% is controlled by 157,700 nonindustrial private forest-land owners with some form of written plan. Ownership size has a strong influence on whether a plan has been prepared: plans are uncommon among small and midsize ownerships, while over half of the owners as well as more than half forest land in the 1,000-plus acre properties are covered by a written plan.

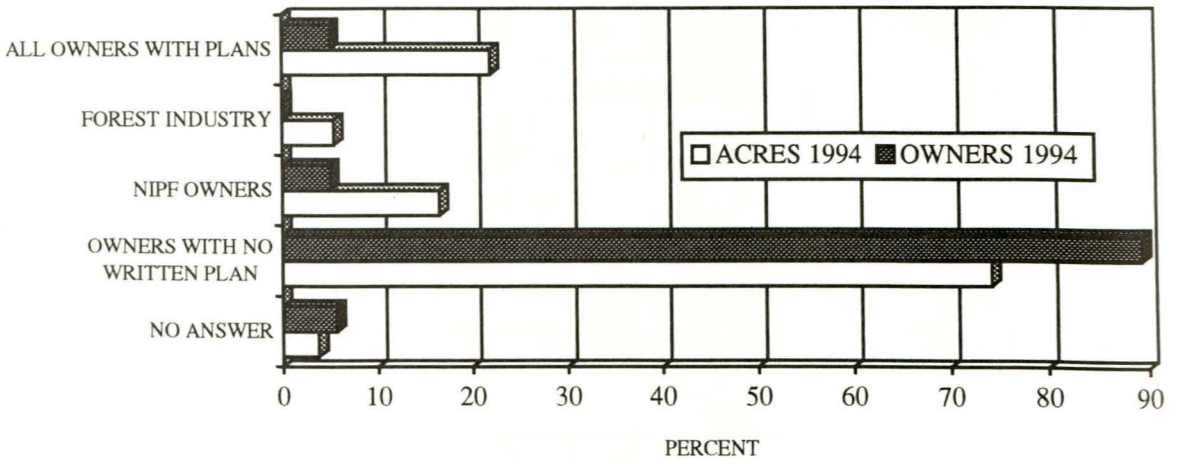


Figure 9. Distribution of private ownerships and acres of forest land owned, by whether a written management plan has been prepared, Central Hardwood Region, 1994.

## CONCLUSIONS

The increasing number of owners makes the job of communicating with owners more difficult. The short tenure of ownership among those who hold smaller tracts will require an ongoing effort to communicate forestry assistance programs. Communication of such programs also is vital to reach the large proportion of forest-land owners and acreage that currently is not covered by a written management plan.

Parcelization of forest land into smaller tracts can be viewed by some to have serious adverse impacts on timber production and wildlife species that require interior forest conditions. Also, additional disturbance no doubt will result from future road construction and installation of utility lines. Still, despite these concerns, more people today are enjoying the forests of the Central Hardwood Region than in 1978, and attitudes about timber harvesting remain positive.

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# IDENTIFYING FOREST LANDS IN URBAN AREAS IN THE CENTRAL HARDWOOD REGION

Thomas W. Birch[1], Rachel Riemann Hershey[2] and Philip Kern[3]

Abstract: Forests in urban areas are an important component of urban and suburban environments. They provide places for recreation and environmental education, wildlife habitat for species adapted to living near humans, contribute to general human physical and psychological health. Knowing how much and what type of forest exists in urban areas provides critical baseline data for evaluating changes in that resource, and critical data to access factors that influence forest management and health. Forest Inventory and Analysis (FIA) data combined with population estimates constitute two valuable sources of information that show promise in understanding the extent, structure, and location of the forest resource in urban areas of the central hardwood region. Ownership information was examined to assess whether size and/or type of ownership is related to population density. There is increasing parcelization of forested lands and an increase in the numbers of nonindustrial private forest-land owners. Identifying where this parcelization is taking place in relation to population centers is important in understanding the process of fragmentation of forest land. The use of these techniques offer an opportunity to monitor changes in this important resource.

## INTRODUCTION

Forested areas are important to the urban and suburban ecosystems of the central hardwood region. They provide sites for recreation, environmental education, and wildlife habitat; improve air and water quality; and enhance our physical and psychological well being. Because forests in urban areas may represent a significant portion of the forest resource in a heavily populated region, knowing their extent, structure and location and being able to monitor changes in these forests can benefit planners and managers. The object of this study is to identify forest land falling within urban areas or closely associated with areas of high human population density.

Many municipalities are involved in issues related to urban forestry, particularly the development of strategies to preserve local forest areas (Ries 1994). Factors such as noise and energy-cost reduction, rain and runoff control, wildlife habitat, recreational areas, and aesthetics must be considered when planning improvements to forests in urban areas (Deneke and Grey 1986). Another consideration is ecological landscaping, for example, the use of trees and shrubs to reduce temperature in urban heat islands. This technique has been effective in cutting energy costs both for cooling in the summer and heating in the winter.

We describe a methodology for identifying forest land in urban areas accurately, consistently, and relatively easily across the central hardwood region. The use of data sources from the USDA Forest Service's Forest Inventory and Analysis (FIA) and the U.S. Bureau of Census provide an opportunity to begin to characterize the forest in "urban" areas. Population and/or household density may be indicators of levels of human influence on forest land that could be useful in developing an understanding of the definition of this urban subset of forest area and where it fits with other uses of the term "urban forest."

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[1]Resource Analyst, USDA Forest Service, Northeastern Forest Experiment Station, Forest Inventory and Analysis, 5 Radnor Corporate Center, Radnor, PA 19086.

[2]Forester/Geographer, USDA Forest Service, Northeastern Forest Experiment Station, Forest Inventory and Analysis, 5 Radnor Corporate Center, Radnor, PA 19086.

[3]Intern, USDA Forest Service, Northeastern Forest Experiment Station Forest Inventory and Analysis, 5 Radnor Corporate Center, Radnor, PA 19086.

In a study of the structure and function of urban forests in the Chicago area, McPherson and others (1994) determined the amount, location, and character of that resource. The results will allow the assessment of its status and change over time. That assessment and inventories conducted for other municipalities have concentrated on trees along streets and in parks that fall within various jurisdictions. While useful at the local level, such inventories do not allow assessment of the status of forests in urban areas at the state or regional level (Nowak 1994). The method used here allows us to evaluate the stand level characteristics and location of the forests in urban areas at the state level. We will use New York and Pennsylvania as examples of data and methods that could be applied in the Central Hardwood Region.

## DATA

The population density and other urban data used in this study are from the 1990 census, collected and distributed by the U.S. Bureau of Census (1990). Population and housing counts are from the Summary Tape File (STF1). Information on population density was calculated as total persons divided by total land area (persons/square mile). Most census data are available at the block or block-group level, which ensures the confidentiality of individual records. Blocks and block groups are census statistical areas, homogeneous in population but varying considerably in size between densely and less densely populated areas. For example, each block group contains an average population of 1,000. Population information used in this study was at the block-group level.

The Bureau of Census identifies areas as "urban" or "rural" at the block level (in NY blocks range in size form 6 to 248,863 acres). An area is considered urban if it consists of a central city or cities and surrounding closely settled territory that together have a population of 50,000 or more. Areas designated as "urban" also include places of 2,500 or more persons (U.S. Bureau Census 1992). The designation is roughly equivalent to a population density of 1,000 persons/square mile plus additional developed areas adjacent to those that are highly populated. However, to discern levels of human influence on forest land, we also used six population-density classes (≥1,000, ≥500 to <1,000, ≥250 to <500, ≥100 to <250, ≥50 to <100, and <50). Urban information and all of the census spatial data used in this study were obtained from 1994 TIGER/Line files (Wyatt and others 1990). Data on population density were obtained from the Land View II CD-ROM (U.S. Environmental Protection Agency and others 1995).

The forest data used in this study were collected by the Northeastern Forest Experiment Station's Forest Inventory and Analysis unit (NE-FIA). The extensive nature of the FIA inventory is suited to summaries and statistics at the state and county level. The forest inventory conducted by NE-FIA is based on a two-tiered, stratified, random sample. The first tier is a grid of 1-acre photo-interpretation (PI) points overlaid on available statewide, small-scale aerial photography. The grid is square with a spacing of approximately 3,900 feet (ground distance) between points, representing approximately 1 acre of 350. Each photo point on the photograph is interpreted using a stereo pair of photographs and a stereoscope, and is classified into a land use (forest or nonforest) and a cubic-foot volume class (if forested). This dataset is one of two used in this study. The second was a FIA plot dataset, a subset of the PI points, which are visited and inventoried for a wider range of attributes and for ground verification of the photo interpretation. The ground sample represents approximately 1 acre in 6,500. The two datasets offer slightly different types of information that describe the forest resource. The first offers greater spatial resolution; the second offers greater attribute resolution.

The locations of FIA plots in New York and Pennsylvania were digitized from aerial photographs registered to U.S. Geological Survey topographic maps. A formal test has not been completed but the locational error of plots in New York has been estimated to be within ± 300 feet. This level of accuracy was more than sufficient for this investigation. There are 5,403 ground locations in New York and 5,298 in Pennsylvania. New York had 95,792 photo plots and Pennsylvania had 88,024. A caution when analyzing FIA plot data for New York is that in this inventory cycle the plots in the Adirondacks that fell on reserved land were not inventoried. Also, no ground plots were measured in the five boroughs of New York City or in Nassau County, as they were allowed to have a nonforest land use. This resulted in a spatial gap in the data set. The photo plots contain no such hole in the spatial distribution across population-density classes. Both types of plots in the Adirondacks would fall primarily in the lowest density class. The opposite is true for New York City and Nassau County for which the population density classes of virtually every area exceeds 1,000/square mile.

## METHODS

We devised a method to identify forests in urban areas from the FIA inventory process based on the Forest Service definition of "forest" (at least 1 acre in size and 10% stocked with trees) from both PI points and plot data, and from Bureau of Census population data and definition of "urban." All four definitions were used in a series of combinations (census urban and plots, census urban and PI points, population density ≥1,000 and plots, and population density ≥1,000 and PI points). We provide an example in New York of how the additional data collected on FIA plots can be used to describe forest stand characteristics (forest type, stand size, stand age, and stocking levels) within urban areas. In Pennsylvania, we performed a preliminary examination of the land uses and ownership characteristics of the forest land in each of the population density classes. A 10% sample of both forest and nonforest ground plots was further divided into land use classes modified from the set used in the Chicago study (McPherson and others 1994) and the amount of tree cover present. Additional information on the ownership of this sample was examined to determine which ownerships were associated with different population density classes. Data from the Pennsylvania ownership study was used for this purpose (Birch and Stelter 1994).

## RESULTS AND DISCUSSION

The results of all four combinations were compared in both New York and Pennsylvania: census urban and FIA plots, census urban and FIA PI points, population density ≥1,000 and FIA plots, and population density ≥1,000 and FIA PI points. The amount of forestland reported as falling within urban or the most highly populated areas (where population density was used) ranged from 1.2 to 3.1% in New York (241,100 to 598,400 acres) (Tables 1 and 2, Figures 1-4). In Pennsylvania, between .95 and 2.8% of the forestland was reported in those categories (177,800 to 430,500 acres) (Tables 5 and 6, Figures 5-8).

Table 1. Plots and PI points in urban and rural population classes, New York

| | Urban | Rural | Off map | Total |
|---|---|---|---|---|
| Number of blocks | 145,436 | 141,425 | 0 | 286,861 |
| Area: 1,000 acres | 2,387 | 29,387 | 0 | 31,774 |
| percent | 8 | 92 | 0 | 100 |
| | Number of plots | | | |
| Forested: number | 93 | 3,006 | 0 | 3,099 |
| percent | 3 | 97 | 0 | 100 |
| Nonforested: number | 278 | 2,025 | 1 | 2,304 |
| percent | 12 | 88 | 0 | 100 |
| Total: number | 371 | 5,031 | 1 | 5,403 |
| percent | 7 | 93 | 0 | 100 |
| | Number of PI points | | | |
| Forested: number | 1,635 | 54,318 | 4 | 55,957 |
| percent | 3 | 97 | 0 | 100 |
| Nonforested: number | 4,759 | 32,870 | 2 | 37,631 |
| percent | 13 | 87 | 0 | 100 |
| Unassigned: number | 202 | 2,002 | 0 | 2,204 |
| percent | 9 | 91 | 0 | 100 |
| Total: number | 6,596 | 89,190 | 6 | 95,792 |
| percent | 7 | 93 | 0 | 100 |

Tables 3-4 and 7-8 present a breakdown of each urban, rural and population density class by forest and nonforest in New York and Pennsylvania, respectively. In these two states, the census-designated urban areas are forested over at least 25% of their area. In NY the urban class has a proportion of 25:75 (forest: nonforest) land. That proportion is slightly lower in the highest population density class (≥1,000) at 16:84, and slightly higher in the next population density class at 36:64. The proportions are very close to 50:50 for the next three population density classes, until it finally switches over to more forest than nonforest at 70:30 in the lowest population density class. Pennsylvania is similar, containing forested land in 22% of the census-designated urban area.

Table 2. Plots and PI points by population-density class, New York

| | Density class | | | | | | |
|---|---|---|---|---|---|---|---|
| | ≥1000 | ≥500 to <1000 | ≥250 to <500 | ≥100 to <250 | ≥50 to <100 | <50 | Total |
| Number of blocks | 12,228 | 629 | 460 | 792 | 676 | 981 | 15,766 |
| Area: 1,000 acres | 1,542 | 718 | 1,075 | 4,240 | 6,907 | 17,291 | 31,733 |
| percent | 5 | 2 | 3 | 13 | 22 | 54 | 100 |
| | Number of plots | | | | | | |
| Forested: number | 38 | 47 | 88 | 403 | 636 | 1,877 | 3,099 |
| percent | 1 | 2 | 3 | 13 | 21 | 61 | 100 |
| Nonforested: number | 205 | 84 | 105 | 431 | 658 | 821 | 2,304 |
| percent | 9 | 4 | 5 | 19 | 29 | 36 | 100 |
| Total: number | 243 | 131 | 193 | 834 | 1,294 | 2,708 | 5,403 |
| percent | 4 | 2 | 4 | 15 | 24 | 50 | 100 |
| | Number of PI points | | | | | | |
| Forested: number | 709 | 724 | 1,367 | 5,563 | 9,965 | 37,629 | 55,957 |
| percent | 1 | 1 | 2 | 10 | 18 | 67 | 100 |
| Nonforested: number | 3,434 | 1,177 | 1,484 | 6,333 | 10,280 | 14,923 | 37,631 |
| percent | 9 | 3 | 4 | 17 | 27 | 40 | 100 |
| Unassigned: number | 128 | 76 | 119 | 359 | 476 | 1,046 | 2,204 |
| percent | 6 | 3 | 5 | 16 | 22 | 47 | 100 |
| Total: number | 4,271 | 1,977 | 2,970 | 12,255 | 20,721 | 53,598 | 95,792 |
| percent | 4 | 2 | 4 | 15 | 24 | 50 | 100 |

Table 3. Proportion of plots and PI points by rural and urban population classes, New York

| | Urban | Rural | Off map | Total |
|---|---|---|---|---|
| | Number of plots | | | |
| Forested: number | 93 | 3,006 | 0 | 3,099 |
| percent | 25 | 60 | 0 | 57 |
| Nonforested: number | 278 | 2,025 | 1 | 2,304 |
| percent | 75 | 40 | 100 | 43 |
| Total: number | 371 | 5,031 | 1 | 5,403 |
| percent | 100 | 100 | 100 | 100 |
| | Number of PI points | | | |
| Forested: number | 1,635 | 54,318 | 4 | 55,957 |
| percent | 25 | 61 | 67 | 59 |
| Nonforested: number | 4,759 | 32,870 | 2 | 37,631 |
| percent | 72 | 37 | 33 | 39 |
| Unassigned: number | 202 | 2,002 | 0 | 2,204 |
| percent | 3 | 2 | 0 | 2 |
| Total: number | 6,596 | 89,190 | 6 | 95,792 |
| percent | 100 | 100 | 100 | 100 |

Table 4. Proportion of Plots and PI points by population-density class, New York

| | Density class | | | | | | Total |
|---|---|---|---|---|---|---|---|
| | ≥1000 | ≥500 to <1000 | ≥250 to <500 | ≥100 to <250 | ≥50 to <100 | <50 | |
| | Number of plots | | | | | | |
| Forested: number | 38 | 47 | 88 | 403 | 636 | 1,877 | 3,099 |
| percent | 16 | 36 | 46 | 48 | 49 | 70 | 57 |
| Nonforested: number | 205 | 84 | 105 | 431 | 658 | 821 | 2,304 |
| percent | 84 | 64 | 54 | 52 | 51 | 30 | 43 |
| Total: number | 243 | 131 | 193 | 834 | 1,294 | 2,708 | 5,403 |
| percent | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | Number of PI points | | | | | | |
| Forested: number | 709 | 724 | 1,367 | 5,563 | 9,965 | 37,629 | 55,957 |
| percent | 17 | 37 | 46 | 45 | 48 | 70 | 59 |
| Nonforested: number | 3,434 | 1,177 | 1,484 | 6,333 | 10,280 | 14,923 | 37,631 |
| percent | 80 | 60 | 50 | 52 | 50 | 28 | 39 |
| Unassigned: number | 128 | 76 | 119 | 359 | 476 | 1,046 | 2,204 |
| percent | 3 | 3 | 4 | 3 | 2 | 2 | 2 |
| Total: number | 4,271 | 1,977 | 2,970 | 12,255 | 20,721 | 53,598 | 95,792 |
| percent | 100 | 100 | 100 | 100 | 100 | 100 | 100 |

Table 5. Plots and PI points in urban and rural population classes, Pennsylvania

| | Urban | Rural | Off map | Total |
|---|---|---|---|---|
| Number of blocks | 145,615 | 172,158 | 37 | 317,810 |
| Area: 1,000 acres | 1,979 | 27,498 | 0.09 | 29,477 |
| percent | 7 | 93 | 0 | 100 |
| | Number of plots | | | |
| Forested: number | 72 | 3,069 | 1 | 3,143 |
| percent | 2 | 98 | 0 | 100 |
| Nonforested: number | 259 | 1,897 | 0 | 2,155 |
| percent | 12 | 88 | 0 | 100 |
| Total: number | 331 | 4,966 | 1 | 5,298 |
| percent | 6 | 94 | 0 | 100 |
| | Number of PI points | | | |
| Forested: number | 1,439 | 49,267 | 4 | 50,710 |
| percent | 3 | 97 | 0 | 100 |
| Nonforested: number | 4,206 | 30,733 | 2 | 34,941 |
| percent | 12 | 88 | 0 | 100 |
| Unassigned: number | 199 | 2,173 | 1 | 2,373 |
| percent | 8 | 92 | 0 | 100 |
| Total: number | 5,844 | 82,173 | 7 | 88,024 |
| percent | 7 | 93 | 0 | 100 |

Table 6. Plots and PI points by population-density class, Pennsylvania.

| | Density class | | | | | | |
|---|---|---|---|---|---|---|---|
| | ≥1000 | ≥500 to <1000 | ≥250 to <500 | ≥100 to <250 | ≥50 to <100 | <50 | Total |
| Number of blocks | 8,049 | 677 | 678 | 1,056 | 663 | 811 | 11,934 |
| Area: 1,000 acres | 1,304 | 695 | 1,688 | 5,202 | 6,324 | 14,260 | 29,477 |
| percent | 4 | 2 | 6 | 18 | 22 | 48 | 100 |
| | Number of plots | | | | | | |
| Forested: number | 30 | 36 | 105 | 402 | 652 | 1,917 | 3,143 |
| percent | 1 | 1 | 3 | 13 | 21 | 61 | 100 |
| Nonforested: number | 190 | 78 | 192 | 560 | 535 | 601 | 2,155 |
| percent | 9 | 4 | 9 | 26 | 25 | 36 | 100 |
| Total: number | 220 | 114 | 297 | 962 | 1,187 | 2,518 | 5,298 |
| percent | 4 | 2 | 6 | 18 | 22 | 48 | 100 |
| | Number of PI points | | | | | | |
| Forested: number | 617 | 600 | 1,690 | 6,421 | 10,339 | 31,043 | 50,710 |
| percent | 1 | 1 | 3 | 13 | 20 | 61 | 100 |
| Nonforested: number | 3,122 | 1,388 | 3,159 | 8,692 | 8,247 | 10,333 | 34,941 |
| percent | 9 | 4 | 9 | 25 | 24 | 29 | 100 |
| Unassigned: number | 110 | 85 | 188 | 533 | 588 | 869 | 2,373 |
| percent | 5 | 4 | 8 | 22 | 25 | 36 | 100 |
| Total: number | 3,849 | 2,073 | 5,037 | 15,646 | 19,174 | 42,245 | 88,024 |
| percent | 4 | 2 | 6 | 18 | 22 | 48 | 100 |

Table 7. Proportion of plots and PI points by rural and urban population classes, New York

| | Urban | Rural | Off map | Total |
|---|---|---|---|---|
| | Number of plots | | | |
| Forested: number | 72 | 3,069 | 1 | 3,143 |
| percent | 22 | 62 | 100 | 59 |
| Nonforested: number | 259 | 1,897 | 0 | 2,156 |
| percent | 78 | 38 | 0 | 41 |
| Total: number | 331 | 4,966 | 1 | 5,298 |
| percent | 100 | 100 | 100 | 100 |
| | Number of PI points | | | |
| Forested: number | 1,439 | 49,267 | 4 | 50,710 |
| percent | 25 | 60 | 57 | 58 |
| Nonforested: number | 4,206 | 30,733 | 2 | 34,941 |
| percent | 72 | 37 | 29 | 40 |
| Unassigned: number | 199 | 2,173 | 1 | 2,373 |
| percent | 3 | 3 | 14 | 2 |
| Total: number | 5,844 | 82,173 | 7 | 88,024 |
| percent | 100 | 100 | 100 | 100 |

Table 8. Proportion of Plots and PI points by population-density class, Pennsylvania

| | Density class | | | | | | |
|---|---|---|---|---|---|---|---|
| | ≥1000 | ≥500 to <1000 | ≥250 to <500 | ≥100 to <250 | ≥50 to 100 | <50 | Total |
| | Number of plots | | | | | | |
| Forested: number | 30 | 36 | 105 | 402 | 652 | 1,917 | 3,143 |
| percent | 14 | 32 | 35 | 42 | 55 | 76 | 59 |
| Nonforested: number | 190 | 78 | 192 | 560 | 535 | 601 | 2,155 |
| percent | 86 | 68 | 65 | 58 | 45 | 24 | 41 |
| Total: number | 220 | 114 | 297 | 962 | 1,187 | 2,518 | 5,298 |
| percent | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| | Number of PI points | | | | | | |
| Forested: number | 617 | 600 | 1,690 | 6,421 | 10,339 | 31,043 | 50,710 |
| percent | 16 | 29 | 33 | 41 | 54 | 74 | 58 |
| Nonforested: number | 3,122 | 1,388 | 3,159 | 8,692 | 8,247 | 10,333 | 34,941 |
| percent | 81 | 67 | 63 | 56 | 43 | 24 | 40 |
| Unassigned: number | 110 | 85 | 188 | 533 | 588 | 869 | 2,373 |
| percent | 3 | 4 | 4 | 3 | 3 | 2 | 2 |
| Total: number | 3,849 | 2,073 | 5,037 | 15,646 | 19,174 | 42,245 | 88,024 |
| percent | 100 | 100 | 100 | 100 | 100 | 100 | 100 |

The advantage with any of the four combinations of plots, PI points, census-designated urban areas and population density is that the data are readily available nationwide. Thus, it is a methodology for identifying forest in urban areas that can be applied relatively easily and consistently throughout the country. And since both data sources exist over time, this method is applicable to both historical and future data, allowing us to track changes in area, composition, and location of urban forests over time. Differences in regional development patterns and predominate land uses mean that there will be regional differences with respect to the types of land use and tree cover that are captured or missed by this definition of forest in urban areas.

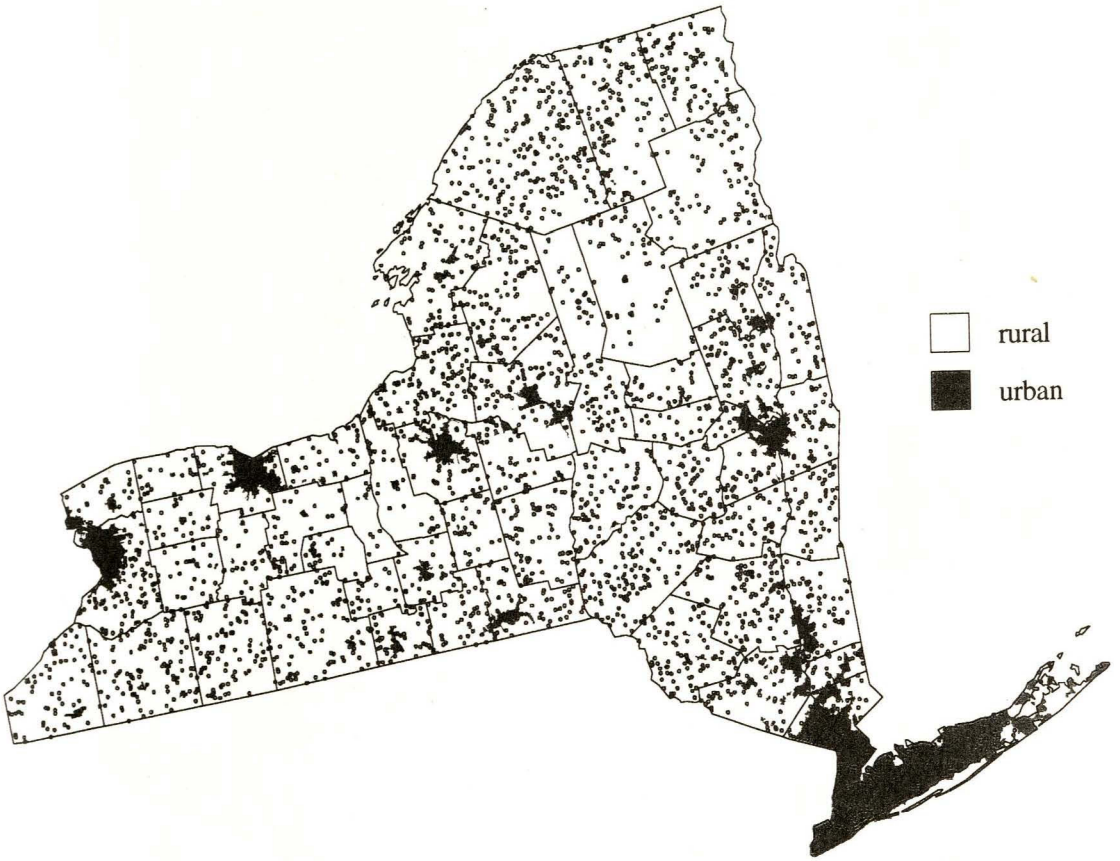


Figure 1. Forested plots and urban/rural classes, New York (population data from the 1990 census; plot data from the 1993 inventory).

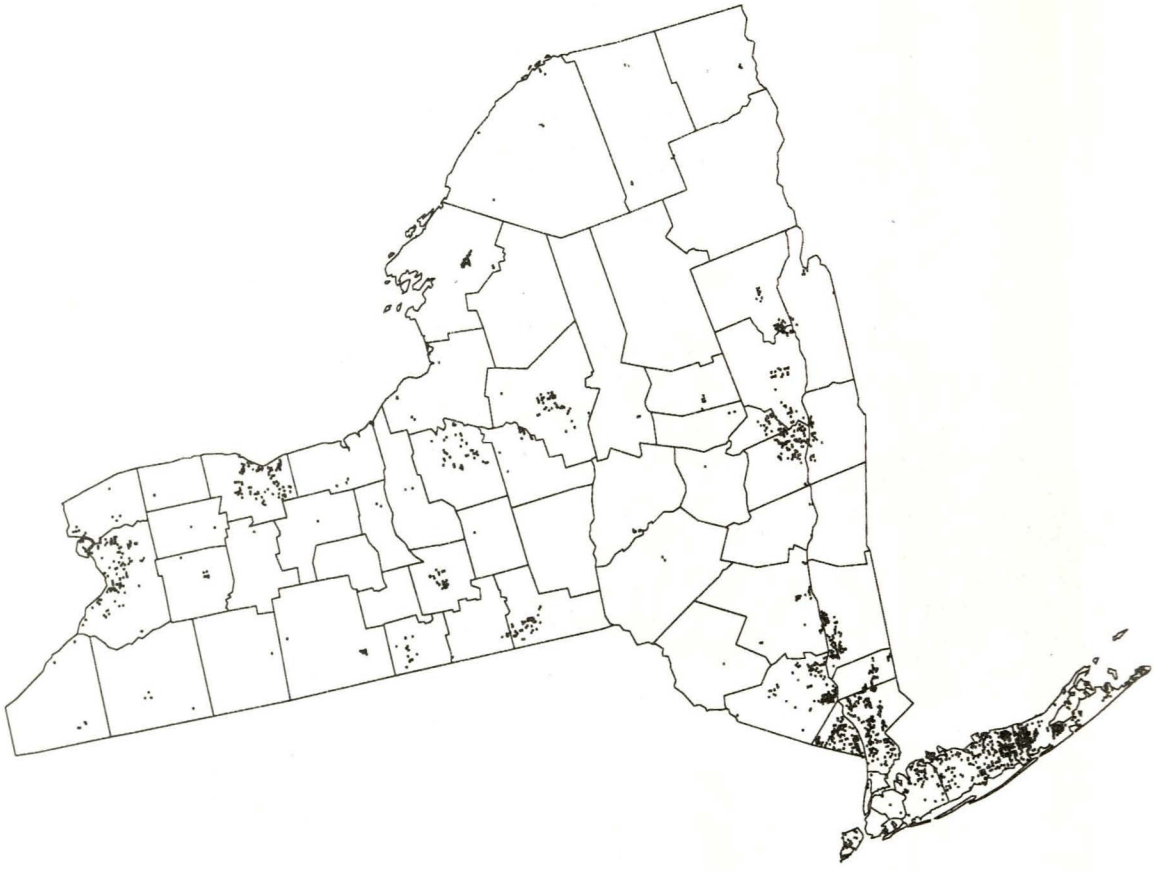

Figure 2. Forested photo-interpetation points in New York within census-designated urban.

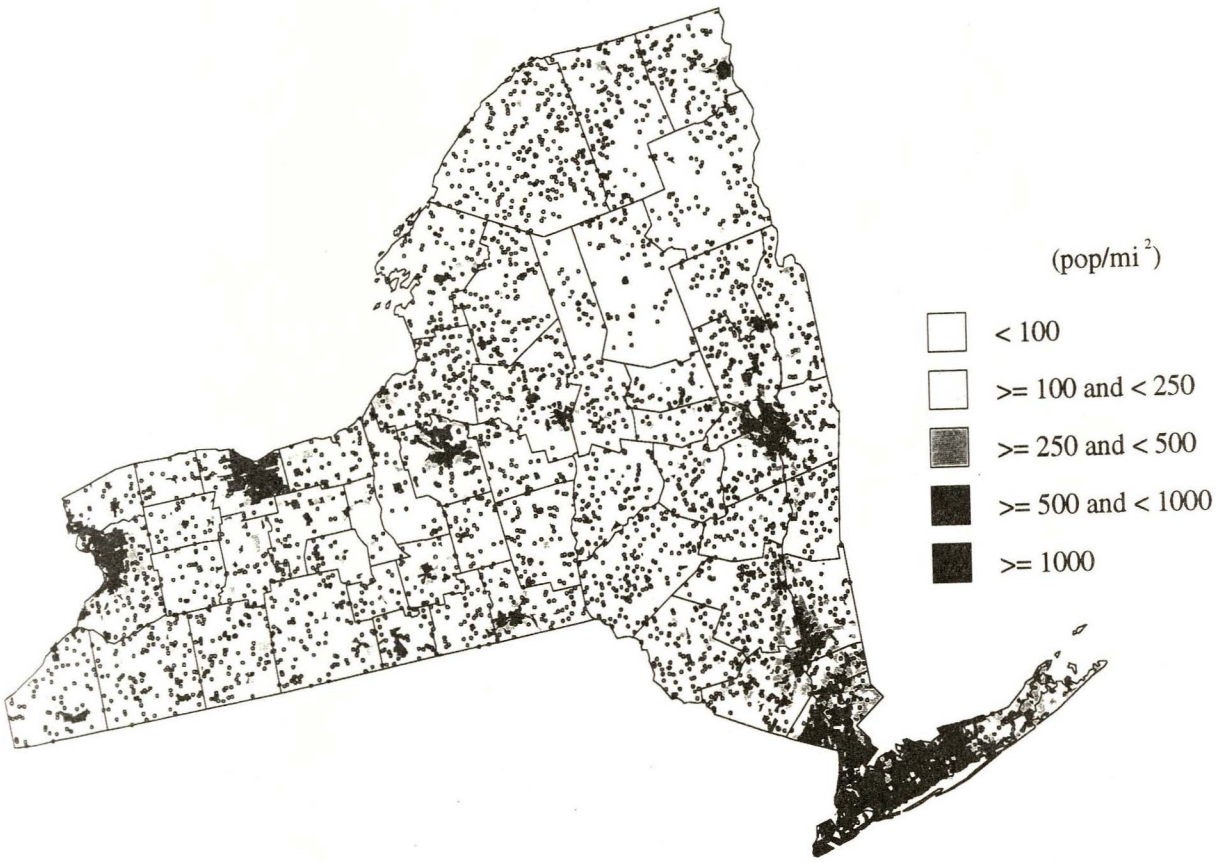


Figure 3. Forested plots and population density classes, New York (population data from the 1990 census; plot data from the 1993 inventory).

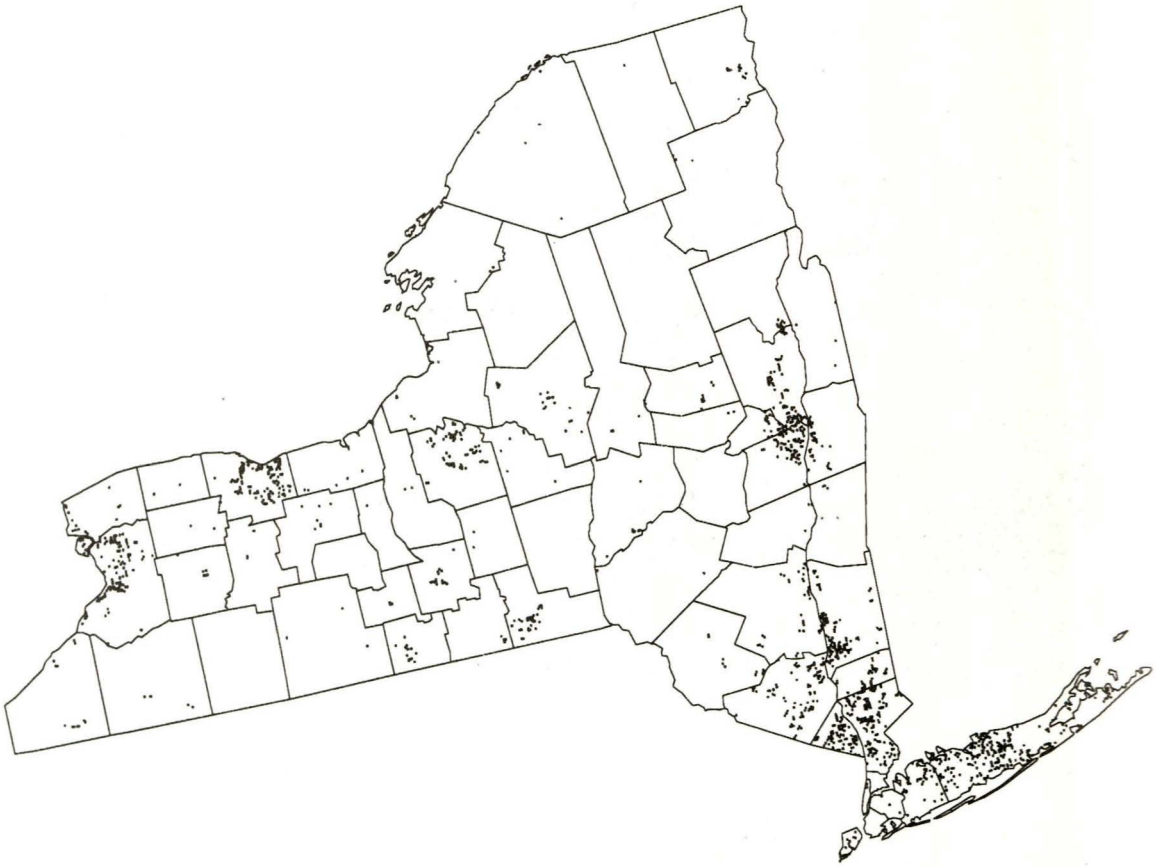

Figure 4. Forested photo-interpretation points in New York within population density class of ≥ 500 persons/mi$^2$.

Figure 5. Forested plots and urban/rural classes, Pennsylvania (population data from the 1990 census; plot data from the 1989 inventory).

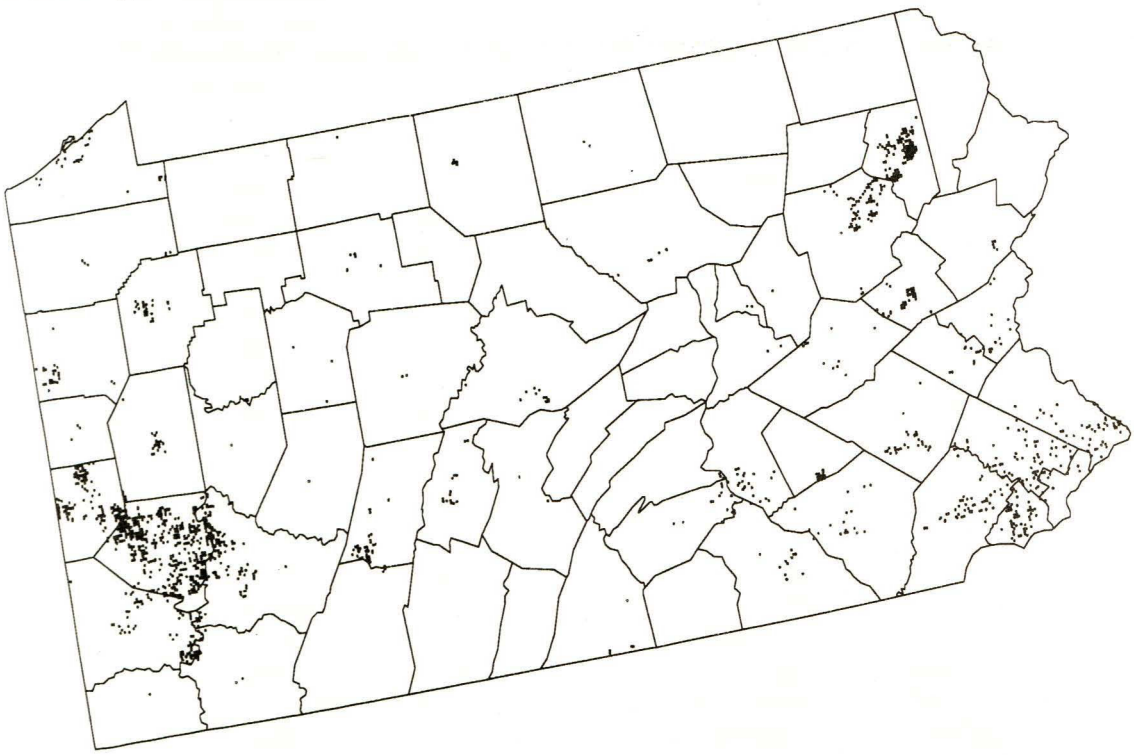

Figure 6. Forested photo-interpretation points in Pennsylvania within census-designated urban.

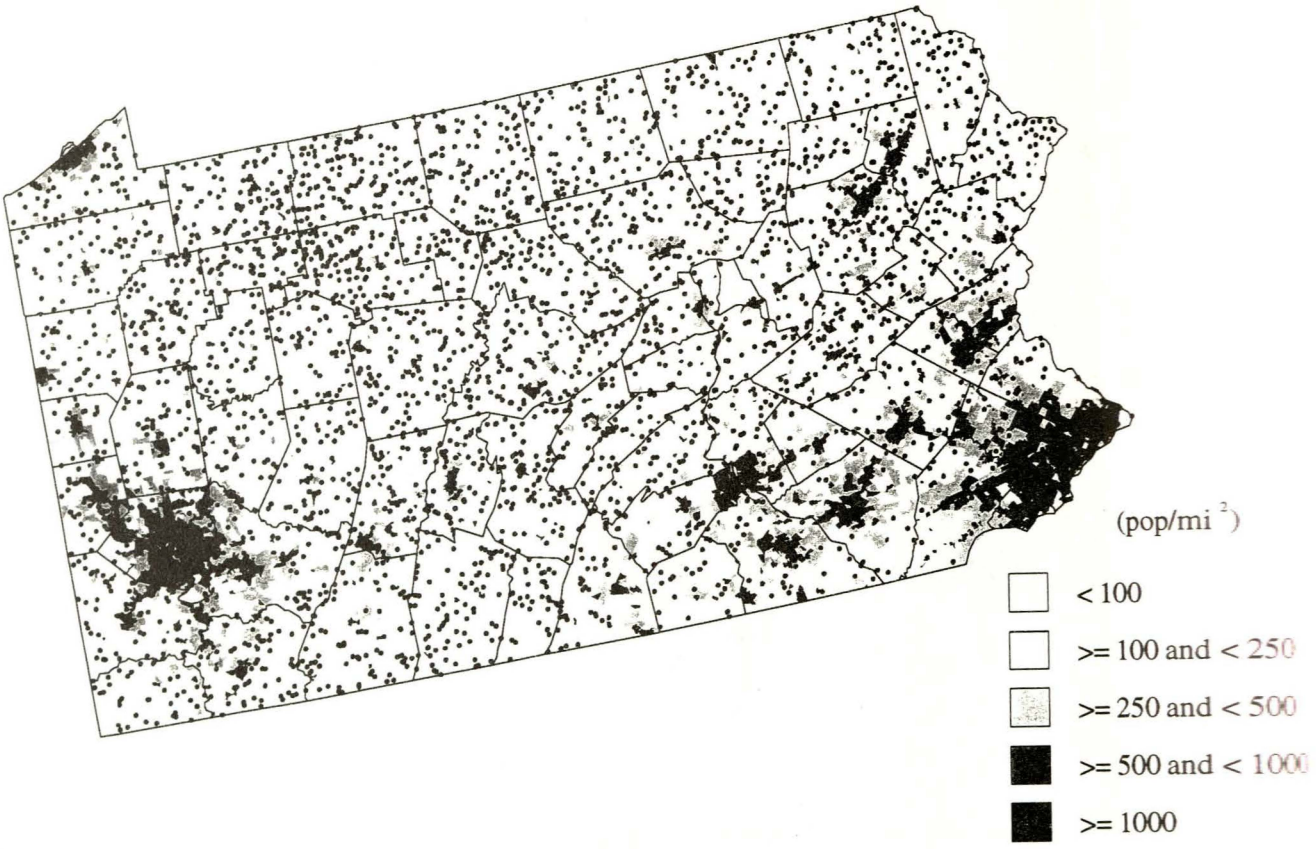


Figure 7. Forested plots and population-density classes, Pennsylvania (population data from the 1990 census; plot data from the 1989 inventory).

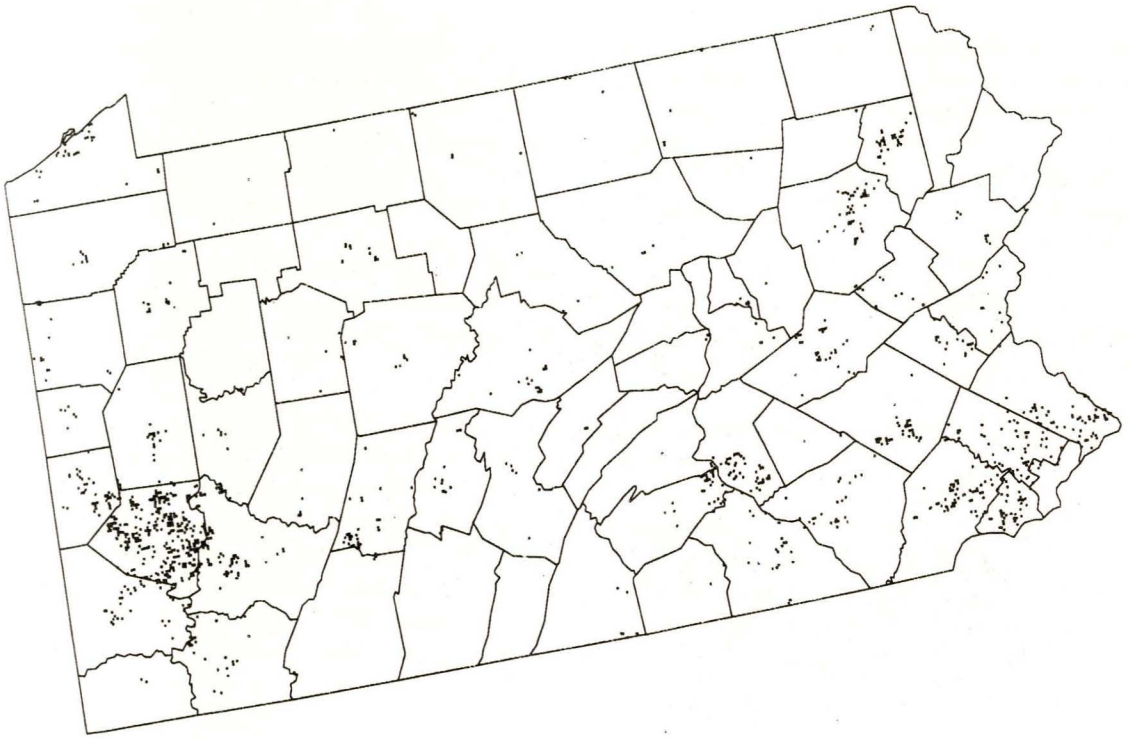

Figure 8. Forested photo-interpretation points in Pennsylvania within population density class of ≥ 500 persons/mi$^2$.

Differences between census-designated urban and population density. The census-designated 'urban' area appears to correspond most closely to a population density class of ≥500 persons/square mile. Although the basis of the definition is areas with a population density of ≥1,000 persons/square mile, all the additional exceptions in the definition effectively incorporate an equivalent amount of land, although in some cases slightly different locations.

Differences between FIA plots and FIA photo interpretation (PI) points. Although some differences might be expected between methods using the PI points to define forest land and those using the plots, because the former is more likely to capture tree cover versus land use class (e.g., any stand with a managed understory, such as mowing, is considered nonforest due to nonforest land use), little difference was in fact observed. This is mostly likely because in those two states the photo interpreters were also trying to interpret for forest/nonforest land use and land cover simultaneously, and were able to match the on-the-ground class fairly closely. In states photo interpreted since 1994, (Maine, Vermont, New Hampshire) points are being interpreted for forest cover first and then land use, allowing the use of PI points as a better source of data on tree cover (both with the Forest Service definition of one acre minimum area and 10% minimum stocking level (forested) and without that minimum (including the category 'nonforest with trees').

The primary difference between the different combinations of 'forest' and 'urban' was not in this case between the amount of forest in 'urban' areas reported, but in the kinds of information offered from that analysis. Using FIA plots allows a further description of the stand characteristics of the forested plots in each urban or population density class. Tables 9-12 illustrate this in New York, by presenting a breakdown of those plots falling within the census-designated urban area by forest type group, stand size, stand age, and stocking level. This allows us to further describe the characteristics of this forest land occurring in these urban areas. In general, the older and larger stands (sawtimber and pole timber stands more than 45 years old) are in the lower Hudson Valley. The younger and smaller stands (sapling stands 0 to 25 years old) are in the corridor that comprises Buffalo, Rochester, and Syracuse. As with any statistical summary, care must be taken when small subsets of the area are examined that contain few sample plots. This situation might occur if individual urban areas were selected for investigation.

Table 9. Distribution of forest types within the urban area, New York

| Forest type group | Percentage of plots |
|---|---|
| Pines | 7.6 |
| Spruce/fir | 2.2 |
| Pitch pine | 12.0 |
| Oak/pine | 4.3 |
| Oak-hickory | 23.9 |
| Elm/ash/red maple | 13.1 |
| Maple/beech/birch | 34.8 |
| Aspen | 2.2 |

Table 10. Distribution of forest stand sizes within the urban area, New York

| Stand size | Percentage of plots |
|---|---|
| Sawtimber | 46.7 |
| Poletimber | 30.4 |
| Seedling/sapling | 22.8 |

Table 11. Distribution of forest stand ages within the urban area, New York

| Stand age (years) | Percentage of plots |
|---|---|
| 0 | 3.3 |
| 25 | 29.3 |
| 45 | 25.0 |
| 65 | 22.8 |
| 85 | 7.6 |
| 105 | 3.3 |
| Uneven-aged stands | 8.7 |

Table 12. Distribution of forest stocking levels within the urban area, New York

| Stocking level | Percentage of plots |
|---|---|
| nonstocked | 1.1 |
| poorly stocked | 13.0 |
| medium stocked | 29.3 |
| fully stocked | 32.6 |
| overstocked | 23.9 |

In contrast to the extensive amount of attribute and descriptive detail offered by the FIA plots, using FIA photo interpretation points allows a better description of exactly where that forestland falling within urban or highly populated areas occurs. As can be seen in Figures 2, 4, 6, and 8, the greater intensity of sample points (approximately 1 for every 300 acres) provides a much clearer picture of where those small, scattered or larger areas of forest within the urban or highly populated areas occurs. The intensity of sampling at the scale of the FIA plots (approximately 1 for every 5,000 acres) is not sufficient to provide this information.

Using census-designated urban offers a very specifically defined and widely recognized and understood definition of urban areas. Using population density (or a similar variable such as household density) provides a means to separate out gradations of probable human use of and influence on the forest—two realities that make the urban forest of such interest for study and monitoring.

These combinations, of FIA plot and PI data with census urban and population density data, do provide a descriptive picture of what subset of the FIA (USFS) forestland falls within urban and highly populated areas. But also of considerable interest is how this reported information compares with the total picture of tree-cover within those areas. Certainly not every element of forests in urban areas is captured and can be described with these inventories. Tree-covered areas less than 1 acre in size or less than 120 feet wide are not included in the FIA plot inventory, nor are certain 'nonforest' land uses such as street trees, or city parks, or trees in residential, commercial, and institutional areas. And although the photo interpretation points in the most recent inventories (starting with New Hampshire) are identifying tree cover separately from land use, in NY and PA the photo interpretation points were largely subject to the same definitions as the FIA plots. This situation does miss many areas with tree cover in urban areas that are often viewed and managed as urban forest. In order to gain a better understanding of how these components are classified or missed under the combination methods used here, additional photo interpretation of a sample of these plots was undertaken in Pennsylvania. An approximately 10% sample (512 of 5,298 plots) was selected across the six population-density classes and photo interpreted again, this time for landuse in the proximity to the one acre sample point. The land-use classes used were a modified version of those in the Chicago study, and included (McPherson and others 1993):

| Land use class | Land cover description |
|---|---|
| Residential | Rural/forested, no yards |
| | Suburban/many trees, clumps |
| | Urban/only a few trees |
| Industrial/commercial/ institutional | Building dominated |
| | Park dominated |
| Transportation | Road and railroad |
| Agricultural | Cropland and pasture land |
| | Areas surrounded by cropland |
| Wild/Vacant | Forests dominate |

The results were compared for each population density category to determine the types of land use being captured as 'forest' or 'nonforest'. Of particular interest are the types of tree cover and land use captured as the forested category, and the types of tree cover placed in the nonforest category—those areas that might be considered "urban forest" (i.e., all tree cover) but are not captured here (Tables 13 and 14). Table 13 shows the distribution of land uses among forested plots by population density class. It can be seen that in the lower population-density areas (fewer than 100 persons/square mile), agricultural and wild lands dominate. In areas with more than 250 persons/square mile, residential lands dominate and the classes begin to capture forested plots associated with industrial parks, institutions, and transportation corridors. Land uses for the nonforest plots are shown in table 14. The population-density classes of fewer than 250 persons/square mile are dominated by agriculture. With increases in density we observed an increase in institutional, industrial, transportation and residential uses. Many of these classes have trees associated with them that many would classify as "urban forest".

Table 13. Distribution of sample plots by population density class for forested plots, Pennsylvania

| Population density class (persons/mi²) | Land use | No. of plots | Percentage in class |
|---|---|---|---|
| <50 | Wild | 158 | 85 |
| | Near agriculture | 24 | 13 |
| | Rural residential | 5 | 2 |
| ≥ 50 to < 100 | Wild | 33 | 52 |
| | Near agriculture | 20 | 32 |
| | Rural residential | 8 | 13 |
| ≥ 100 to < 250 | Wild | 20 | 51 |
| | Near agriculture | 7 | 18 |
| | Rural residential | 11 | 28 |
| | Suburban residential | 1 | 3 |
| ≥ 250 to < 500 | Near agriculture | 2 | 18 |
| | Rural residential | 4 | 37 |
| | Suburban residential | 1 | 9 |
| | Urban residential | 2 | 18 |
| | Industrial | 1 | 9 |
| | Institutional | 1 | 9 |
| ≥ 500 to < 1,000 | Wild | 1 | 25 |
| | Near agriculture | 1 | 25 |
| | Rural residential | 1 | 25 |
| | Suburban residential | 1 | 25 |
| ≥ 1,000 | Suburban residential | 2 | 67 |
| | Transportation corridor | 1 | 33 |

Ownership information was examined to assess whether size and/or type of ownership is related to population density. Trends in ownership influence the process of parcelization and fragmentation, forest management and forest use of forest land. We observed that most of the publicly owned forest at the state and national level was in density classes of fewer than 100 persons/square mile. Also, forest industry lands in Pennsylvania occur in less densely populated portions of the state (Birch 1994). Small forest ownerships were more closely associated with population densities greater than 250 persons/square mile. For additional information on the definitions and process of parcelization and fragmentation of the forest in New York, see Birch (1996).

Table 14. Distribution of sample plots by population density class for nonforest plots, Pennsylvania

| Population density class (persons/mi²) | Land use | No. of plots | Percentage in class |
|---|---|---|---|
| <50 per square mile | Agricultural | 33 | 60 |
| | Rural residential | 12 | 22 |
| | Vacant | 5 | 9 |
| | Quarry | 1 | 2 |
| | Transportation | 3 | 5 |
| ≥ 50 to < 100 | Agricultural | 34 | 67 |
| | Rural residential | 2 | 4 |
| | Suburban residential | 3 | 6 |
| | Vacant | 3 | 6 |
| | Quarry | 3 | 6 |
| | Transportation | 3 | 6 |
| | Industrial | 2 | 4 |
| | Institutional | 1 | 2 |
| ≥ 100 to < 250 | Agricultural | 30 | 54 |
| | Rural residential | 5 | 9 |
| | Suburban residential | 9 | 17 |
| | Urban residential | 1 | 2 |
| | Vacant | 1 | 2 |
| | Industrial | 3 | 5 |
| | Transportation corridor | 2 | 3 |
| ≥ 250 to < 500 | Agricultural | 6 | 32 |
| | Rural residential | 4 | 21 |
| | Suburban residential | 6 | 32 |
| | Vacant | 2 | 10 |
| | Industrial | 1 | 5 |
| ≥ 500 to < 1,000 | Suburban residential | 4 | 57 |
| | Agricultural | 2 | 29 |
| | Institutional | 1 | 14 |
| ≥ 1,000 | Rural residential | 2 | 11 |
| | Suburban residential | 5 | 28 |
| | Urban residential | 9 | 50 |
| | Institutional | 1 | 6 |

## CONCLUSION

Different definitions of urban forest produce different estimates but also reveal important information. Using the Bureau of Census definition of urban has several advantages. It is widely recognized and understood, it is specifically defined, and it can be applied across the country. However, these data do not identify forests outside of census-designated urban areas that also are heavily influenced by human activity. Analyzing where forested plots and photo points fall within different population-density classes allows us to identify other subsets of forest land that may be subject to significant human use and influence.

As mentioned earlier, both FIA ground plots and PI points offer two valuable types of information that describe the forest resource. Each ground plot allows ground verification of the photo interpretation and a large number of variables that provide greater attribute resolution. For example, plots can be described in terms of stand age, stand size, and species composition. Photo interpretation points hold little attribute information but have a much greater spatial resolution, providing an opportunity to better understand where the urban forest occurs.

We present a technique for estimating and describing the forest in urban areas of two of the larger and heavily populated states of the Northeast. We need to apply this technique across the Northeast and in other regions of the country. As new inventories are completed by FIA, we will have an opportunity to monitor changes in this valuable resource.

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# DISTRIBUTION AND EXTENT OF TREE MORTALITY IN NORTH CENTRAL HARDWOOD FORESTS

J. Michael Vasievich, Sharon L. Hobrla, and Mark H. Hansen[1]

Abstract: Forest inventory data shows that biophysical agents and human causes account for annual losses of more than a half-billion $ft^3$ of timber in North Central hardwood forests. This paper reports on an analysis of forest inventory data to determine the extent and distribution of tree mortality in four forest types–Aspen-Birch, Elm-Ash-Cottonwood, Maple-Beech-Birch and Oak-Hickory. Eight states were included–Michigan, Minnesota, Wisconsin, Indiana, Illinois, Iowa, Missouri, and Kansas. Plot and tree measurement records were examined to characterize tree mortality; estimate the area affected by occurrences of higher-than-average mortality; to evaluate the proportion of timber lost to various causal agents and to explain differences in mortality due to location and stand conditions.

Results show that 1% of growing stock volume and 25% of growing stock gross growth are lost each year to all natural and human causes. Mortality was highest for the Aspen-Birch forest type which averaged 30% of growth (14.1 $ft^3$/ac/yr) and lowest for Oak-Hickory which averaged 22% of growth (6.5 $ft^3$/ac/yr). Estimates of the frequency of large losses show that Aspen-Birch had the greatest chance, 8.9%, of having annual mortality exceed 40 cubic feet annually and Oak-Hickory had the lowest chance at 1.4%. Curves are presented that describe the chance of exceeding a certain level of mortality for each forest type and state. Analysis of plot data showed that annual mortality was statistically related to site index, stand age, basal area stocking, and growing stock percent. Models produced adjusted R-square values of 0.36 to 0.91 for all but one model. Basal area stocking was significant and positive in almost all models. Stand age was significant and positive in more than half of the models. Cause of death determinations are highly subjective, so most mortality was classified as other or unknown. In cases where cause of death was determined, disease was generally the most common agent cited. Other major causes of mortality included fire, weather, suppression, insects, and logging. These results provide information to compare mortality with growth and inventory for different hardwood types across the study region.

[1] The authors are Project Leader and Statistician, respectively, Forest Economics Research, USDA Forest Service, North Central Forest Experiment Station (NCFES), 1407 S. Harrison Road, East Lansing, MI 48823 and Research Forester, NCFES, 1992 Folwell Avenue, St. Paul, MN 55108. We would like to acknowledge assistance provided by Pat Miles, NCFES, St. Paul, MN and support for this work by Frank Thompson and Steve Shifley, Central Hardwoods Ecosystem Management Project, NCFES, Columbia, MO.

# HARDWOOD TIMBER SALES ON STATE FORESTS IN INDIANA:

# CHARACTERISTICS INFLUENCING COSTS AND PRICES

J. Michael Vasievich, W. L. Mills, Jr. and Heidi R. Cherry[1]

Abstract: Timber sales conducted on State-owned forests in Indiana from 1982 to 1994 were analyzed to determine changes in costs and prices and the effect of sale conditions on costs and prices. The data set included 445 sales that ranged in size from less than 1 acre to more than 500 acres. Sales were predominantly partial cuts in mature hardwood timber. Marked timber volume was less than 5 Mbf (thousand board feet) per acre for most sales. Some sales included other methods, such as clearcutting or seedtree harvests, on some or all of the sale area.

Direct costs for planning and conducting timber sales averaged $8.19 per Mbf, excluding overhead. This represented 3.9% of the high bid. Direct sale costs per Mbf increased only slightly from 1982 to 1994, about 0.1% faster than inflation over the period. Bid prices averaged $181 per Mbf over the time period and rose by 3.77% faster than inflation.

Linear regression was used to identify sale characteristics that significantly affect costs and prices. Analysis showed that much of the variability in direct costs and bid prices per Mbf could be explained by available information for each sale. The r-square values were 0.74 for direct costs and 0.91 for high bid. Sale date, contract length, percent preferred species, total number of trees sold, and number of trees sold per acre all increased direct costs per Mbf for conducting sales. Total sale volume and volume per acre were negatively correlated with costs, indicating economies of scale. For stumpage prices, sale date, total sale volume, and percent preferred species were positively correlated. Size of sales in acres, number of culls, and number of trees in the sale reduced bid prices per Mbf. Although not a useful predictor of price, the number of bids was positively correlated with per unit price.

## INTRODUCTION

Marketing and harvesting timber is one of the most important tools used to achieve forest management goals (Callahan and others 1979, Flick 1985). The market value of timber is influenced by many factors, including market conditions, location, site conditions, type and quality of timber offered, and conditions of the sale. Knowledge of how these factors affect the market value of particular timber offerings can be used to determine fair market values and set minimum acceptable bids, estimate bid prices prior to sale, and identify strategies to craft more profitable timber sale offerings. Understanding how changes in harvesting and management practices can affect sale price is important to increased profitability of forest management investments. Information on the costs of managing timber is essential for cost-effective harvesting and management decisions, yet few published sources provide information on the costs of selling timber or describe the relationship between stand characteristics and timber sale economics. This information is particularly lacking for the high-value timber found growing in Central Hardwood forests where selective harvesting provides the financial means to achieve desired species mixes, accomplish regeneration objectives, and sustain productivity. Models developed for mixed sales in Wisconsin (Buongiorno and Young 1984) provide a starting point for this approach.

Most experienced foresters who regularly sell timber are aware of the many factors that influence timber sale values. However, most do not have quantitative estimates of these factors, and their effect on both the costs of conducting

[1]The authors are Project Leader, North Central Forest Experiment Station (NCFES), USDA Forest Service, 1407 S. Harrison Road, East Lansing, MI 48823; Professor of Forestry, Purdue University, West Lafayette, IN 47907; and Forester, NCFES, East Lansing, MI.

timber sales and the resulting timber prices. In this paper, we describe an empirical analysis of the costs and revenues derived from mixed hardwood timber sales in Indiana. Our approach follows hedonic price theory, which indicates that the market value of goods such as timber are determined by their characteristics (Prescott and Puttock 1990, Puttock and others 1990, Rosen 1974). This approach, as we have applied it, considers how different attributes of a timber sale, such as volume, quality, and sale conditions, contribute to the cost of conducting sales and market price. This method, sometimes termed statistical transaction appraisal (Vasievich and others 1988), uses statistical procedures to quantify the relationship between sale characteristics and prices for a set of actual timber sale transactions.

Our primary objective was to describe the characteristics of a typical set of Central Hardwood timber sales and to quantify the effect of timber sale factors and site conditions on the costs of conducting timber sales and the prices that are paid for those sales.

## DATA AND METHODS

Data for this study were taken from 445 individual timber sale reports from State Forests in Indiana[2] as shown in Figure 1. The timber sale reports (Indiana Department of Natural Resources form SF-200) contained information on timber sale characteristics, volume of timber sold by species, and resources used and costs associated with conducting each timber sale such as tree marking, boundary marking, sale preparation, advertising, and showing the sale. Also included was information on the number of bids and the price offered by each bidder. The sale reports contained a general description of the purposes and management objectives for the sale and factors that were considered limitations.

The 445 sales available for analysis were offered from mid-1982 to mid-1994 as shown in Table 1. Sales were sold by competitive bid, and most represented partial harvests of mixed hardwoods (Figure 2) with individual trees marked for cutting. Although specific timber sale objectives differed, information contained on the sale forms indicated that common goals were to sustain and increase stand productivity through the harvest of mature trees and to remove undesired trees. Sales ranged in size from 1 to more than 500 acres. Timber volume sold ranged from less than 1 Mbf per acre to more than 25 Mbf per acre. Several sales had very high volumes per acre, but all these were small – less than 3 acres. Timber sales composed primarily of cut logs were excluded from analysis as were sales with missing data and outliers that represented extremely high priced sales. We recorded information on sale date, size, volume and number of trees sold, volume of preferred species, length of cutting contract, number of bids, high and low bid values, and various costs for administering the sale. We also reviewed comments on marking objectives and silvicultural methods. This information helped us identify factors that could affect costs and prices.

[2] We gratefully acknowledge the assistance of Dr. Burnell C. Fischer and staff of the Indiana Division of Forestry, Indianapolis, IN for providing timber sale reports and additional information required for this study.

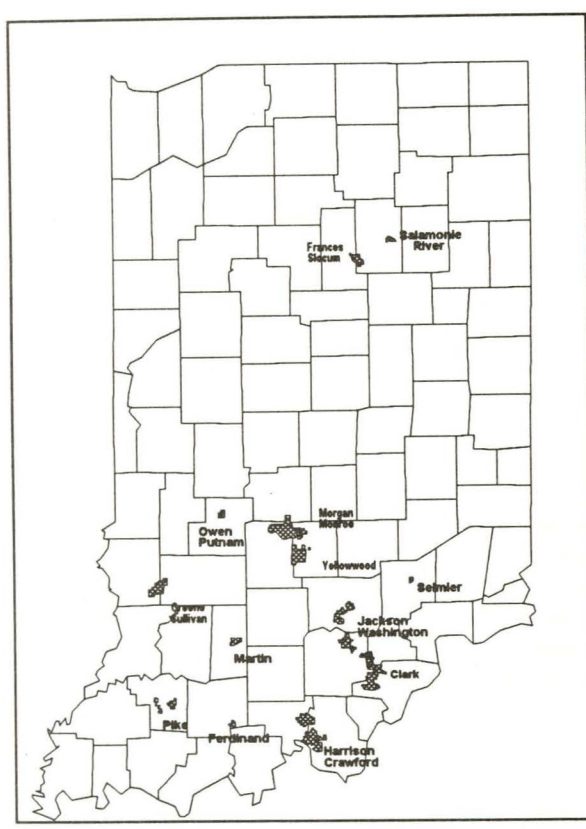


Figure 1. Location of State Forest lands in Indiana included in timber sale cost analysis.

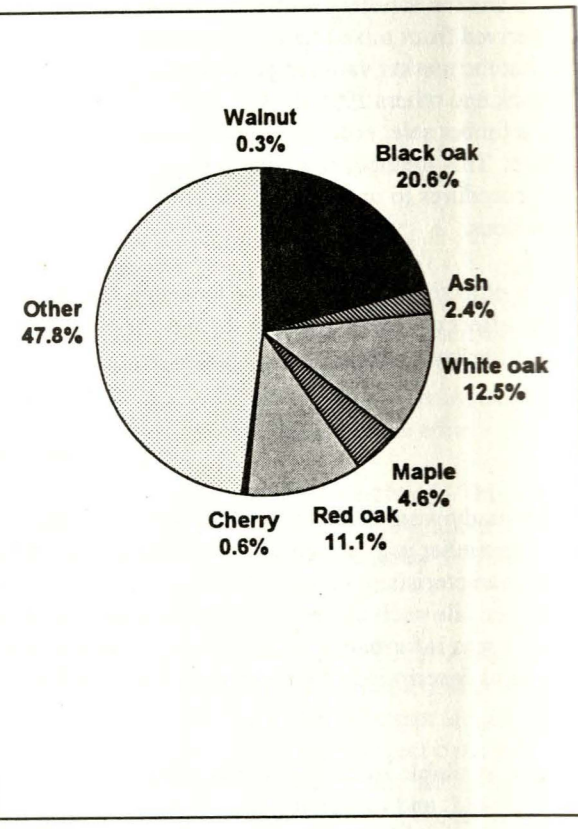


Figure 2. Distribution of timber volume sold, by species.

Table 1. Number, area, and volume sold of 445 hardwood timber sales in Indiana by State Forest

| Forest | Sales | Sale area | | | Volume sold | | |
|---|---|---|---|---|---|---|---|
| | | Average | Minimum | Maximum | Average | Minimum | Maximum |
| | Number | ------- Acres ------- | | | ------- Mbf ------- | | |
| Clark | 32 | 101.3 | 12 | 359 | 149.9 | 11.0 | 338.0 |
| Deam Lake | 1 | 13.0 | 13 | 13 | 23.0 | 23.0 | 23.0 |
| Ferdinand | 33 | 60.6 | 1 | 240 | 109.6 | 13.0 | 377.0 |
| Frances Slocum | 5 | 49.2 | 25 | 80 | 68.6 | 40.0 | 137.0 |
| Greene Sullivan | 3 | 28.3 | 25 | 32 | 107.3 | 75.0 | 150.0 |
| Harrison-Crawford | 46 | 98.1 | 5 | 537 | 123.7 | 6.0 | 334.0 |
| Jackson-Washington | 45 | 64.6 | 2 | 209 | 111.5 | 2.0 | 408.0 |
| Martin | 30 | 68.0 | 2 | 158 | 119.7 | 3.0 | 245.0 |
| Morgan Monroe | 78 | 96.1 | 7 | 213 | 154.6 | 18.0 | 378.0 |
| Owen Putnam | 29 | 73.4 | 14 | 200 | 120.4 | 33.0 | 223.0 |
| Pike | 18 | 45.9 | 1 | 88 | 70.5 | 22.0 | 165.0 |
| Salmonie | 3 | 50.0 | 20 | 100 | 68.7 | 37.0 | 116.0 |
| Selmier | 3 | 38.3 | 16 | 58 | 95.0 | 60.0 | 132.0 |
| Wyandot | 4 | 115.0 | 33 | 280 | 77.0 | 58.0 | 108.0 |
| Yellowwood | 115 | 67.2 | 3 | 294 | 117.9 | 8.0 | 411.0 |
| All sales | 445 | 76.3 | 1 | 537 | 122.6 | 2.0 | 411.0 |

Timber sales averaged 76.3 acres in size with 122.6 Mbf (Table 2). Sale volume averaged 2.0 Mbf per acre and approximately 206 board feet per tree. The distribution of volume sold per acre is shown in Figure 3, and the distribution of sizes of sales is shown in Figure 4. Fifty-two percent of all volume sold over the period was represented by seven species we considered preferred for market purposes. These included walnut, red oak, white oak, white ash, black cherry, maple, and black oak. More than 40 other hardwood and softwood species made up the remaining volume, including various hickories, butternut, other oaks, yellow poplar, beech, sycamore, sweetgum, elms, bigtooth aspen, sassafras, basswood, cottonwood, hackberry, persimmon, river birch, other ashes, soft maples, black locust, honey locust, black gum, buckeye, and Kentucky coffee tree. Softwoods included on sales were white, Virginia, and red pines, and Eastern redcedar. The sale reports also identified the volume of timber in each sale considered prime—the best veneer quality of desired species. This averaged less than 1% of the total sale volume for all sales.

Table 2. Average timber sale characteristics for 445 timber sales on Indiana State forests

| Year | Sales | Sale area | Volume | Volume | Trees sold | Average volume | Preferred species | Prime timber | Cutting period |
|---|---|---|---|---|---|---|---|---|---|
| | Number | Acres | Mbf | Mbf/ac | Number | bf/tree | % | % | Months |
| 1982 | 14 | 58.0 | 89.6 | 1.6 | 436 | 205 | 45.0 | 1.4 | 11.1 |
| 1983 | 36 | 98.3 | 131.5 | 1.8 | 617 | 213 | 48.4 | 1.0 | 11.2 |
| 1984 | 35 | 81.6 | 118.1 | 1.7 | 554 | 213 | 55.9 | 0.4 | 11.2 |
| 1985 | 44 | 84.7 | 135.5 | 1.9 | 691 | 196 | 53.0 | 0.5 | 11.9 |
| 1986 | 52 | 77.0 | 136.0 | 2.2 | 679 | 200 | 49.9 | 0.4 | 12.3 |
| 1987 | 45 | 69.4 | 132.8 | 2.0 | 635 | 209 | 48.2 | 0.2 | 11.3 |
| 1988 | 42 | 70.5 | 119.6 | 3.0 | 557 | 215 | 52.0 | 0.5 | 11.8 |
| 1989 | 40 | 61.0 | 112.7 | 2.5 | 583 | 193 | 47.9 | 0.4 | 11.4 |
| 1990 | 41 | 77.7 | 111.5 | 1.6 | 527 | 212 | 49.8 | 0.4 | 11.5 |
| 1991 | 37 | 88.2 | 123.9 | 1.9 | 611 | 203 | 44.8 | 0.4 | 10.8 |
| 1992 | 26 | 62.6 | 107.8 | 1.7 | 501 | 215 | 51.8 | 0.6 | 11.7 |
| 1993 | 23 | 67.3 | 108.7 | 1.7 | 465 | 234 | 53.0 | 2.6 | 12.0 |
| 1994 | 10 | 87.1 | 145.8 | 1.7 | 776 | 188 | 59.5 | 1.4 | 12.5 |
| All | 445 | 76.3 | 122.6 | 2.0 | 594 | 206 | 50.3 | 0.6 | 11.6 |

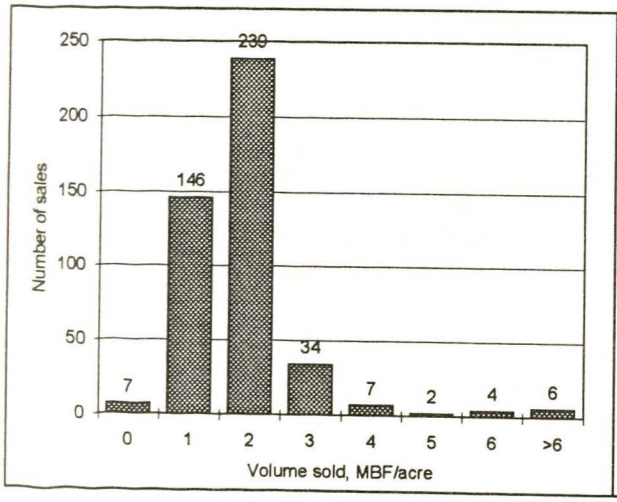


Figure 3. Distribution of timber sales by volume per acre.

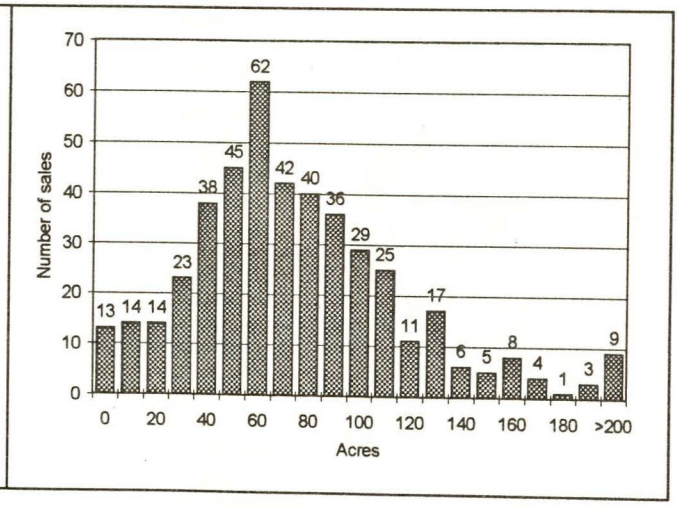


Figure 4. Distribution of timber sales by size of sale.

## Timber Sale Costs and Revenues

Timber sale bids ranged from less than $50 per Mbf to more than $500 per Mbf and averaged $181 per Mbf for all sales as shown in Figure 5. The average timber sale was sold for $22,209 for 122.6 Mbf. The average sale attracted 3.3 bids. High bids averaged 175% of low bids (Table 3).

Table 3. Average timber sale revenues and volume characteristics for 445 mixed hardwood timber sales in Indiana

| Year | Sales | Volume sold | Volume sold | Sale price | Unit price | Bids | Bid spread |
|---|---|---|---|---|---|---|---|
| | Number | Mbf | Mbf/ac | $/sale | $/Mbf | Number | % |
| 1982 | 14 | 89.6 | 1.6 | 12,050 | 126 | 3.2 | 164 |
| 1983 | 36 | 131.5 | 1.8 | 19,666 | 135 | 3.4 | 162 |
| 1984 | 35 | 118.1 | 1.7 | 18,598 | 152 | 2.9 | 139 |
| 1985 | 44 | 135.5 | 1.9 | 16,938 | 130 | 2.9 | 154 |
| 1986 | 52 | 136.0 | 2.2 | 19,329 | 138 | 3.1 | 156 |
| 1987 | 45 | 132.8 | 2.0 | 23,323 | 173 | 4.0 | 167 |
| 1988 | 42 | 119.6 | 3.0 | 26,831 | 214 | 3.4 | 168 |
| 1989 | 40 | 112.7 | 2.5 | 24,701 | 213 | 4.6 | 355 |
| 1990 | 41 | 111.5 | 1.6 | 20,559 | 192 | 3.0 | 160 |
| 1991 | 37 | 123.9 | 1.9 | 19,146 | 161 | 3.6 | 171 |
| 1992 | 26 | 107.8 | 1.7 | 27,884 | 258 | 3.7 | 165 |
| 1993 | 23 | 108.7 | 1.7 | 30,752 | 272 | 2.3 | 128 |
| 1994 | 10 | 145.8 | 1.7 | 45,688 | 347 | 2.4 | 142 |
| All | 445 | 122.6 | 2.0 | 22,209 | 181 | 3.3 | 175 |

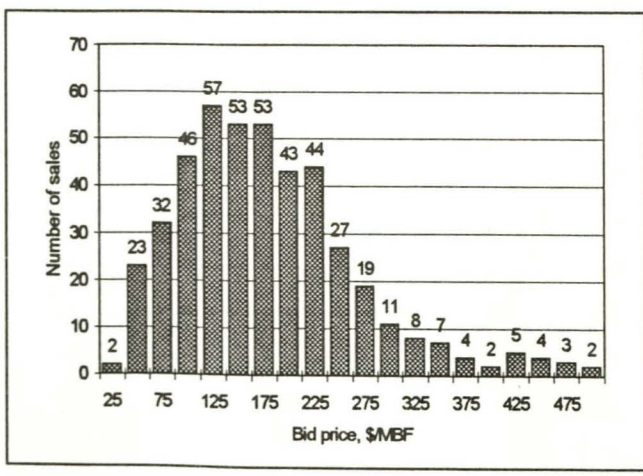


Figure 5. Distribution of timber sales by price per Mbf.

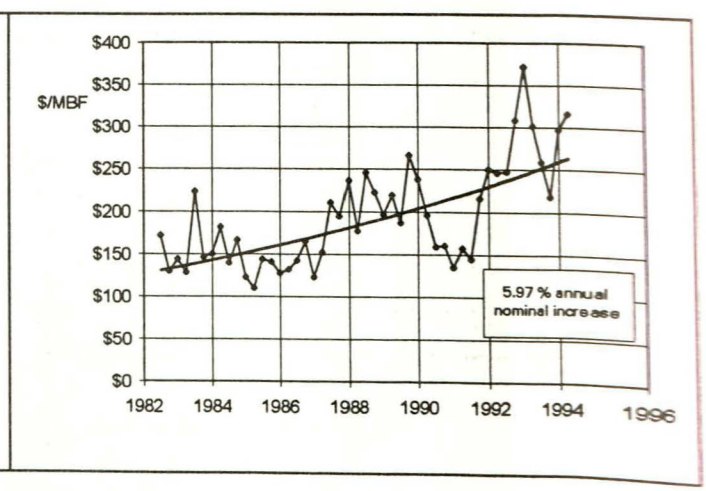


Figure 6. Average bid price for all sales by year and quarter in $/Mbf.

Average total costs for conducting timber sales were $1,348 per sale, or about $11 per Mbf (Table 4). Most sales cost less than $15 per Mbf, but a few were more costly, as indicated in Figure 7. Direct costs represented 64% of total sale costs or about $869 per sale on average. Other costs, mostly related to road construction, maintenance, and site improvements, added $479 per sale. Timber marking made up the greatest proportion of total costs, 71% of all direct costs, as shown in Table 5 and Figure 8. Other components included paperwork at 11% of direct costs; boundary marking, 7%; advertising, 5%; and showing the sale, 6%.

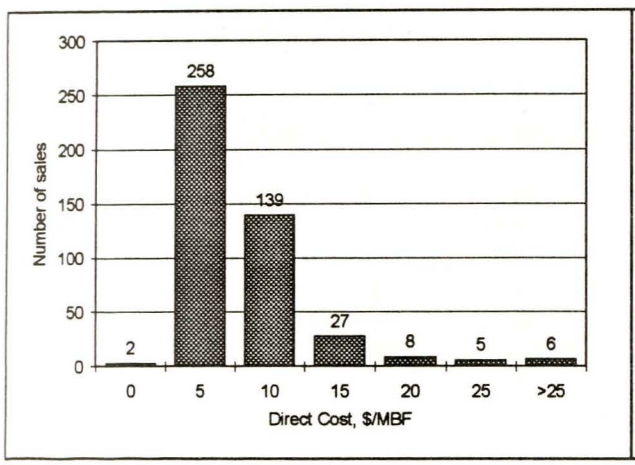


Figure 7. Distribution of sales by cost in $/Mbf.

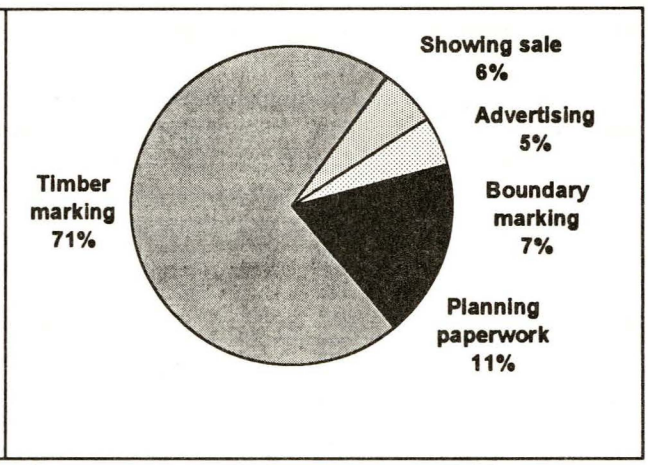


Figure 8. Distribution of sale costs by activity.

Table 4. Average timber sale costs for 445 mixed hardwood sales in Indiana

| Year | Sales | Sale area | Total costs | Direct costs | Direct costs | Direct costs |
|---|---|---|---|---|---|---|
| | Number | Acres | $/sale | $/sale | $/Mbf | % of sale |
| 1982 | 14 | 58.0 | 926.86 | 682.11 | 13.93 | 5.66 |
| 1983 | 36 | 98.3 | 1,113.70 | 833.12 | 7.82 | 4.24 |
| 1984 | 35 | 81.6 | 1,003.50 | 819.43 | 9.15 | 4.41 |
| 1985 | 44 | 84.7 | 1,011.54 | 874.83 | 6.96 | 5.17 |
| 1986 | 52 | 77.0 | 1,040.42 | 852.41 | 7.50 | 4.41 |
| 1987 | 45 | 69.4 | 1,491.71 | 1,004.50 | 7.72 | 4.31 |
| 1988 | 42 | 70.5 | 1,338.80 | 905.12 | 9.19 | 3.37 |
| 1989 | 40 | 61.0 | 1,422.05 | 783.70 | 8.44 | 3.17 |
| 1990 | 41 | 77.7 | 1,317.05 | 795.52 | 8.05 | 3.87 |
| 1991 | 37 | 88.2 | 1,950.97 | 930.51 | 7.42 | 4.86 |
| 1992 | 26 | 62.6 | 1,607.93 | 870.86 | 8.11 | 3.12 |
| 1993 | 23 | 67.3 | 1,430.14 | 817.47 | 7.50 | 2.66 |
| 1994 | 10 | 87.1 | 3,202.67 | 1,256.15 | 9.40 | 2.75 |
| All | 445 | 76.3 | 1,348.20 | 868.89 | 8.19 | 3.91 |

Table 5. Average costs by timber sale activity

| Year | Sales | Planning paperwork | Boundary marking | Timber marking | Advertising | Showing sale | Other costs |
|---|---|---|---|---|---|---|---|
| | Number | $/sale | | | | | |
| 1982 | 14 | 359 | 48 | 53 | 64 | 41 | 245 |
| 1983 | 36 | 557 | 58 | 58 | 96 | 50 | 281 |
| 1984 | 35 | 561 | 57 | 52 | 98 | 51 | 184 |
| 1985 | 44 | 634 | 38 | 54 | 93 | 56 | 137 |
| 1986 | 52 | 602 | 59 | 47 | 88 | 55 | 188 |
| 1987 | 45 | 688 | 71 | 40 | 102 | 102 | 487 |
| 1988 | 42 | 614 | 60 | 53 | 97 | 81 | 434 |
| 1989 | 40 | 540 | 68 | 51 | 101 | 37 | 625 |
| 1990 | 41 | 464 | 50 | 39 | 78 | 60 | 522 |
| 1991 | 37 | 602 | 45 | 40 | 101 | 83 | 1,020 |
| 1992 | 26 | 633 | 49 | 47 | 107 | 35 | 737 |
| 1993 | 23 | 563 | 39 | 58 | 95 | 64 | 613 |
| 1994 | 10 | 915 | 28 | 67 | 148 | 99 | 1,947 |
| All | 445 | 589 | 54 | 49 | 96 | 63 | 478 |

Rates of Change for Costs and Prices

Costs and prices reported for the set of timber sales were analyzed to determine rates of change for the period mid-1982 to mid-1994. Linear regression methods were used to fit a log-transform model—Ln(price) = a + b(year)—to determine the annual rate of change in prices. Selected values were also adjusted for inflation with the Producer Price Index. Results are shown in Table 6. Producer prices rose by 2.2% per year over the analysis period. During the same time, average timber prices increased at a compound annual rate of 5.97% and 3.77% faster than inflation as indicated in Figure 6. Total sale costs rose by 6.09% per year after adjusting for inflation. Direct costs (i.e., planning, marking, showing, and advertising the sale, excluding other costs) increased by only 0.11% per year over inflation, indicating that other costs included on the sale report increased substantially faster than inflation.

Table 6. Rates of price change for timber sale costs and bid prices, before and after adjusting for inflation

| | Nominal rate of value change | Real rate of value change |
|---|---|---|
| | %/year | %/year |
| Bid price, $/Mbf | 5.97 | 3.77 |
| Total sale cost, $/Mbf | 6.09 | 3.89 |
| Direct sale cost, $/Mbf | 2.31 | 0.11 |
| Producer Price Index | 2.20 | NA |

Effects Of Sale Conditions On Costs, Prices, And Number Of Bids

Stepwise regression was used to fit linear models based on sale characteristics for effects on costs, bid price, and number of bids received. Seven models were fit for four dependent sale variables: total sale cost in $/Mbf (Total cost); direct sale costs, (excluding other costs) in $/Mbf (Direct cost); high bid price in $/Mbf (High bid); and number of bids (Bids). Models fit with intercepts did not perform as well as models without intercepts and are not reported.

We tested two measures of sale cost—total cost and direct costs. The total cost model included extraordinary costs that generally represented site or road improvements or maintenance. For the direct cost model, we considered only the costs directly associated with setting up and administering each sale.

Separate price models were fit for high bid without adjusting for inflation (referred to as nominal in Table 7) and after adjusting for inflation with the producer price index (referred to as real in Table 7). These high bid models were also fit with the number of bids included and excluded as an independent variable. The number of bids is not known prior to sale closure, so this variable would not normally be available for predictive models. We hypothesized that additional competition as reflected by a higher number of bids would lead to higher market prices. So, we included the number of bids for real and nominal price models to test the effect of added competition on price.

The final dependent variable, the number of bids, was estimated to identify those sale characteristics that increased sale attractiveness for buyers and contributed to higher levels of competition.

We tested 11 sale characteristics, listed below, as independent variables. We did not have information on some variables, such as access or distance to markets, that we expect would also be useful price determinants.

| | |
|---|---|
| ACRES | Acres in the sale |
| CULLS | Number of cull trees included in the sale |
| CUTLEN | Length of cutting contract, months |
| MBFVOL | Total sale volume in Mbf |
| MONTHS80 | Date of the sale as number of months since January 1, 1980 |
| PCTPRSP | Percent of sale volume in preferred species (walnut, red, white, and black oak, ash, cherry, and hard maple) |
| TREES | Number of trees included in the sale |
| TREESAC | Number of trees sold per acre |
| VOLACRE | Sale volume in Mbf/acre |
| PCTPRIME | Percent of sale volume identified on sale report as prime |
| BIDS | Number of bids |

We selected these variables because they were likely to affect the effort or costs associated with preparing the sale or harvesting timber, because they represented practical economic aspects of operating timber sales, and because most were measured continuous values (as compared to categorical data) and were well documented on the sale forms. We expected the number of culls to reduce average timber quality and be negatively correlated with bid prices. Higher volumes give buyers greater harvesting efficiencies and reduce average transaction costs per unit volume for the buyer. Higher sale volumes also lead to higher marking costs. So, we expected volume sold to be positively correlated with price, but negatively correlated with costs. We expected and found some interactions between variables too. For example, the number of trees offered in a sale was highly correlated with total sale volume. But more trees for the same volume indicated smaller timber and higher logging costs. We found that the number of trees in a sale was negatively related to price, all else being constant.

Results of our statistical analyses are shown in Table 7. Simple linear models performed quite well and resulted in coefficients of determination (r-square) values ranging from 0.71 to 0.93. The signs and magnitudes of the parameter estimates provide some useful insights into the timber sale process and market dynamics. All variables, except PCTPRIME were significant in one or more models. Three variables, CULLS, MBFVOL, and PCTPRSP were significant in all models. Variables were retained in models if they were significant at the 0.10 level.

Table 7. Parameter estimates for models of costs, bid price, and number of bids based on sale characteristics variables only

| Independent Variable | Total costs | Direct costs | High bid (nominal ) | High bid (real ) | High bid (nominal) | High bid (real) | Bids |
|---|---|---|---|---|---|---|---|
| | $/Mbf | $/Mbf | $/Mbf | $/Mbf | $/Mbf | $/Mbf | Number |
| ACRES | | | -0.27697 | -0.24563 | -0.25610 | -0.19601 | |
| CULLS | 0.00867 | 0.00417 | -0.09663 | -0.08461 | -0.08110 | -0.06691 | -0.00179 |
| CUTLEN | 0.02866 | 0.01922 | | | | | 0.00192 |
| MBFVOL | -0.07149 | -0.04868 | 0.53018 | 0.48583 | 0.38913 | 0.33403 | 0.00720 |
| MONTHS80 | 0.06373 | 0.02043 | 0.91657 | 0.62803 | 0.87551 | 0.55163 | |
| PCTPRSP | -0.03060 | 0.03060 | 2.11453 | 2.12859 | 1.66378 | 1.68779 | 0.03715 |
| TREES | 0.00661 | 0.00427 | -0.06783 | -0.05483 | -0.05776 | -0.04827 | |
| TREESAC | | 0.08476 | | | | | 0.09820 |
| VOLACRE | | -0.43446 | | | -2.20020 | | |
| BIDS | | | | | 11.51625 | 11.47334 | |
| R-square | 0.71 | 0.74 | 0.91 | 0.92 | 0.92 | 0.93 | 0.79 |
| MSE | 60.74 | 24.32 | 3512.07 | 2629.50 | 3107.41 | 2216.60 | 3.14 |

We included a variable for time in the model—MONTH80—the date of the sale represented as the number of months since January 1980. This variable served as a simple linear trend term to account for increases in costs and prices over time. MONTH80 was significant and positive for all cost and price models.

Two models were fit to describe sale costs—total costs that included other additional costs mostly related to road access and site improvements and direct costs. The direct cost measure included only costs for marking, showing, advertising, and planning the sale. The other cost component added a significant amount of variability. The model fit for direct costs was substantially better than for total costs and had a higher r-square as well as a mean-squared error (MSE) of less than half of the total cost model. Total sale volume was negatively correlated with costs in $/Mbf indicating the significance of economies of scale. Each additional thousand board feet in the sale reduced average direct costs by 4.9 cents and total costs by 7.1 cents. The coefficient for volume per acre was also negative, indicating lower direct costs for sales with higher volumes per acre. Some sales represented clearcuts, diameter limit cuts, or creation of openings on part or all of the area, and these had generally lower sale administration costs. The coefficient for percent of volume in preferred species (PCTPRSP) was positive for direct costs only. This effect—higher average costs—may indicate greater attention and marking effort for sales with higher amounts of the most valuable species. The coefficient for MONTH80 was positive and significant for all cost and price models, reflecting a general trend of increasing costs and prices.

Two models were fit for bid price—one without the number of bids and one with the number. The number of bids was significant and positive indicating a strong effect due to competition. Each additional bid added approximately $11.50 to the high bid per Mbf. However, including the number of bids increased the model fit only slightly, pushing r-square up by about 0.01. This was not surprising because we found that much of the variability associated with number of bids (as evidenced by the bid model) was explained by independent variables already included in the price models. Percent prime timber volume in the sale was not significant for any model. Percent of the sale in preferred species was positive and added more than $2.12 per Mbf to the high bid for each 1% increase in the combined seven preferred species. Number of trees in the sale was negatively correlated with bid price. This may be due to possibly smaller average tree size in sales with higher numbers of trees, but this has not been otherwise tested. Number of culls in the sale also showed a negative coefficient for all price models. A large number of culls is often seen by buyers as an indicator of lower quality or defect in trees with volume leading to lower value. Volume per acre was significant with a negative coefficient in the nominal price regression that included the number of bids. Interpretation of this negative effect is not clear and may be due to interactions between several independent variables or specific characteristics of this set of timber sales.

A model was fit for the number of bids to analyze sales that attract more competition. The model fit resulted in an r-square of 0.79. Sale volume, percent preferred species, trees per acre, and cutting contract length were all positive, indicating that sales with higher volumes, more trees, higher proportions of the most desirable species, and longer timeframes attract more bidders. Number of culls had a negative coefficient, indicating that including more culls in a sale would reduce the number of potential buyers.

We tested subjective management variables such as the harvesting method, management objective, and cutting restrictions as independent dummy variables (0/1). Models with these variables produced inconclusive results and are not reported in detail here. Several difficulties were apparent. Many measures of cutting method or sale limitations are highly subjective, and we could not effectively differentiate among sales. In most cases, sales included multiple silvicultural objectives, cutting methods, and limitations. We could not determine the extent to which any single method or limitation affected most sales.

Some interesting findings did result, but are inconclusive largely because most sales included a mixture of conditions and methods. Direct costs were $5 less per Mbf for sales where clearcutting was indicated as one of the harvesting methods used for the sale. Selection thinning and seedtree cuts were positively correlated with total costs and direct costs, respectively. These sales require more timber marking effort and increase costs by an estimated $6 to $10 per Mbf. The greatest effect of these variables on bid prices was due to salvage cuts. The high-bid price for salvage cuts was from $48 to $56 lower per Mbf. Several cutting methods and limitation variables appeared in the model for number of bids. We found that substantially more (2.5) bidders were, on average, willing to bid on sales where the seedtree method was indicated. Fewer bidders were interested in sales with poor quality timber or pre-salvage sales. Also, more bidders were interested in sales we classified as having an age restriction—either young or overmature. Sales of overmature timber may have larger and more valuable timber, on average. Although these observations can not be statistically verified for these sales, they do suggest some additional areas for consideration for other analyses.

## SUMMARY

Our analysis of 445 Central Hardwood timber sales on State Forests in Indiana provides insights into the dynamics of timber markets operating primarily in southern Indiana. These forests are managed by the Indiana Department of Natural Resources to produce hardwood sawtimber. Most sales involved partial cutting that favored release of crop trees, systematic removal of culls, and harvest of mature sawtimber trees. Sales averaged 76.3 acres with 122.6 Mbf. The average tree had 206 board feet. Fifty-two percent of the volume sold was in seven preferred species we considered preferred: white, red, and black oak; white ash; black cherry; hard maple; and walnut.

The average sale was for $22,209 and $181 per Mbf. We found that prices for hardwood sawtimber generally increased in both real and nominal terms over the 1982 to 1994 time period, but that the price path demonstrated typical increases and decreases we have observed in hardwood sawtimber markets in other areas. Prices for these high-quality hardwood sawtimber sales increased almost 4% faster than inflation. Sales attracted an average of 3.3 bids each, and the high bid was 175% of the low bid for the typical sale.

Direct costs for conducting these timber sales averaged $8.19 per Mbf and $869 per sale. This included labor and materials for planning and administering the sales, but excluded overhead costs for forest administration and post-sale follow-up. Hence, these figures are likely to underestimate the total costs for conducting these sales by as much as 25 to 40% (our estimate). Additional costs for improvements and maintenance items added another $479 in cost to each sale. Most direct costs, about 71%, were for timber marking, a key management action requiring skilled personnel.

We statistically analyzed 11 site and sale conditions to determine their effect on sale costs, bid prices and number of bids. We were able to explain most of the variability in per unit costs and prices with simple linear models. Model coefficients indicate the comparative effects of sale conditions on cost, price, and number of bids. For total and direct costs in $/Mbf, we found that eight variables were statistically significant at the 0.10 level. R-square was 0.71 for the total cost model and 0.74 for the direct cost model. Six independent variables were significant for all price

models. Acres, number of culls, and number of trees had negative coefficients. Volume sold, sale date, and percent of volume in preferred species had positive coefficients. R-square for price models ranged from 0.91 to 0.93. Models describing real and nominal bid prices performed similarly. We found that sales with higher numbers of bids contributed significantly to higher prices and that each additional bidder added $11.50 per Mbf to the high bid price.

We estimated the number of bids as a function of number of culls (negative), cutting contract length, sale volume, sale date, percent preferred species, and trees per acre. R-square was 0.79 for this model.

Although we have not verified the predictive ability of these models, they are useful in judging the relative value of alternative timber sale configurations. We expect that additional analysis of these sales to refine model formulations would yield more useful predictive models. This analysis involved timber sales conducted only on State Forests in Indiana. We caution readers that these results, while illustrative of timber markets in one place, are not necessarily applicable to other ownerships or locations.

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# THE EFFICACY OF ECONOMIC-DEVELOPMENT PROGRAMS IN FOREST-DEPENDENT COMMUNITIES

Rory Fraser[1]

Abstract: West Virginia is one of the poorest states in the nation and its rural communities are more hard pressed than their urban counterparts. At the same time, West Virginia is one of the most densely forested states in the USA. The combination of poverty amidst a wealth of forest suggest that economic development based on the forest resource could provide solutions for poor rural communities. However, even though decades of public programs have significantly improved the well-being of West Virginia's citizens, rural counties still tend to have the highest unemployment rates, poverty rates and out-migration (Isserman 1995).

There is very little information about the success of forest-based economic development efforts relative to the success of other types of economic development efforts. There is also very little information about the relative success of economic development efforts in forest-dependent communities as compared to other rural communities. This research project will increase our knowledge of the impact of economic development activities on rural communities. The findings is of relevance to West Virginia University's Division of Forestry, West Virginia Council for Community and Economic Development, and the United States Department of Agriculture, Forest Service programs.

The objective of this study is to determine whether economic development programs are effective in forest-dependent-communities. In making this determination, data from the forests, the forest industry, the state park system, the hunters and fishmen as well as and rural communities have been collected and analyzed.

Preliminary results indicate that the combined 1992 economic output from West Virginia's forest-based economic activities was $4.30 billion revenue, 59,510 jobs, and $1.259 billion in wages. The Wood Products Industry contributed $2.135 billion revenue, 16,803 jobs, and $0.465 billion wages. Tourism contributed $1.99 billion revenue, 39,327 jobs, and $0.742 billion in wages. Finally, in-state fishing and hunting activities accounted for $178 million revenue, 3,380 jobs, and $51 million in wages.

Heavily populated, highly forested counties with many campsites, many hotels, high levels of locally owned public land are more likely to generate jobs and revenues in the tourism sector. On the other hand, counties which generate the largest number of wood industry jobs tend to be the larger counties in which major areas are owned by the U.S. Forest Service or the forest industry.

In the next stage of this McIntire-Stennis funded project, the role of economic development initiatives in the growth and development of the forest sector, will be examined.

[1] Division of Forestry, West Virginia University, Morgantown, WV 26506.

# AN EVALUATION OF UNEVEN-AGED CUTTING METHODS IN EVEN-AGED OAK-HICKORY STANDS IN THE BOSTON MOUNTAINS OF ARKANSAS

David L. Graney and Paul A. Murphy[1]

Abstract: A test of group-selection and single-tree selection cutting methods was installed in 80-year-old even-aged oak-hickory stands in the Boston Mountains of northern Arkansas. Twenty-four 11-ac study plots were installed in well stocked stands representing north or east and south or west aspects. Stands between group openings were cut to residual basal areas of 65 and 85 $ft^2$ using free thinning or structural control. This paper describes (1) the effect of aspect on preharvest stand structure and species composition, (2) preharvest reproduction composition, and (3) the effect of cutting method and density treatments on residual stand structure. The implications of existing stand structure and cutting method efficacy are also discussed.

## INTRODUCTION

Public concern over the dramatic visual impact of clearcutting has stimulated interest in alternative forest management systems for upland oak forests in the Midsouth. Acceptable alternatives should avoid the negative visual impacts of clearcutting and must provide the biological conditions necessary for regenerating and maintaining the oaks and other valuable hardwood species. While modifications of shelterwood cutting (Sander and Graney 1993) satisfy the visual and biological requirements, it suffers a "guilt of association" by being an even-aged method. Critics of clearcutting and other even-aged methods have suggested uneven-aged silviculture cutting methods (group selection or single-tree selection) to meet visual and biological objectives.

Uneven-aged cutting methods are designed to create and maintain at least three age classes within the stand. In single-tree selection, all trees marked for cutting are selected for removal based upon their individual merit. But in group selection, the trees removed in regeneration cuts are selected as a group or aggregate, not as individuals; however, the trees removed between the group openings are selected on individual merit. Group selection can be considered a variant of single-tree selection. Periodic cuts are required in both methods to (1) establish and develop reproduction, (2) improve stand structure and quality, and (3) control residual stocking for sustained yield. They differ in how the periodic cuts are made and their effect on species composition. Recent papers (Miller and others 1995, Murphy and others 1993) have provided an excellent description of both methods and have discussed the advantages and disadvantages of each.

Conventional wisdom has considered group selection to be more suitable than single-tree selection for shade-intolerant species, such as the oaks and other desirable species on upland sites in the Midsouth. Group-selection openings of at least 0.4 ac or with diameters at least two times the height of mature dominant or codominant trees in the stand provide the light conditions necessary for the intermediate and shade-intolerant species to compete successfully. However, as with the even-aged methods, successful development of oaks in group openings requires adequate numbers of large advance reproduction when the overstory is removed. A major problem on mesic or productive upland sites is that large oak advance reproduction is often absent or inadequate.

The single-tree selection method has been assumed to be inappropriate for managing upland oaks and other intermediate and shade-intolerant species (Sander and Clark 1971, Sander and Graney 1993). Canopy openings created in single-tree selection cutting have been considered to be too small to allow enough light for oak reproduction, but

[1] Research Foresters, Southern Research Station, USDA Forest Service, 726 W. Maple Street, Fayetteville, AR 72701, and P.O. Box 3516, Monticello, AR 71656-3516.

they do provide sufficient light for a tolerant understory to develop. While existing ecological and silvicultural literature corroborates these assertions, these studies were conducted in mesic ecosystems and included no understory or competition control treatments to favor development and accumulation of large oak reproduction.

However, the single-tree selection system has been successfully adapted to more shade-intolerant species, such as the loblolly-shortleaf pine type (*Pinus taeda* L. and *P. echinata* Mill.), by following certain principles and practices: (1) choosing a species or a species mix that is moderately tolerant in the seedling stage, (2) maintaining a residual basal area low enough to permit the establishment and development of desired reproduction, (3) selecting a cutting cycle to keep basal area levels within acceptable ranges, (4) eliminating overstory and midstory non-target competition by harvesting or manual and chemical treatments, and (5) periodically controlling understory non-target competition (usually with selective herbicides). The combination of a low basal area (one that does not exceed 75 $ft^2$/ac at any time during the cutting cycle) and competition control results in an environment favorable for development of pine regeneration.

Johnson (1992) describes xeric and dry mesic upland sites in the Missouri Ozarks where oak reproduction accumulates over time, possibly providing a pool of large oak reproduction that would sustain an uneven-aged management system. A large industrial forest in the Missouri Ozarks has used a version of single-tree selection on xeric and dry mesic upland sites for more than 40 years, and the method appears to be maintaining a healthy, sustainable forest (Loewenstein and others 1995).

A study has been installed to evaluate effectiveness of group-selection and single-tree selection methods in mature even-aged oak-hickory stands on dry mesic and mesic upland sites in the Boston Mountains of northern Arkansas. The specific objectives are:

(1) To test the feasibility of using group-selection and single-tree selection methods to convert even-aged oak-hickory stands to uneven-aged ones.
(2) To test two methods of regulation and two density levels in combination with group selection.
(3) To compare growth and yield of stands that are managed and regenerated under group selection, two methods of regulation, and two density levels.
(4) To measure and evaluate the development of desirable reproduction in group openings and the establishment and growth of desirable reproduction in the residual stand.
(5) To evaluate the establishment and growth of desirable reproduction with different competition control intensities, overstory densities, aspects, and types of regulation.

Objective (5) is being undertaken as an adjunct to the main study that covers the first four objectives. The main study is called the regulation study, and the adjunct is called the competition study.

In this paper, we describe the preharvest conditions and how the cutting methods and density treatments affected stand structure in the regulation study.

## METHODS

### Study Region

The Boston Mountains are the highest and most southern member of the Ozark Plateaus physiographic province (Figure 1). They form a band 30 to 40 miles wide and 200 miles long from north-central Arkansas westward into eastern Oklahoma. Elevations range from about 900 ft in the valley bottoms to 2,500 ft at the highest point. The plateau is sharply dissected, and most ridges are flat to gently rolling and are generally less than 0.5 mile wide. Mountainsides consist of alternating steep simple slopes and gently sloping benches.

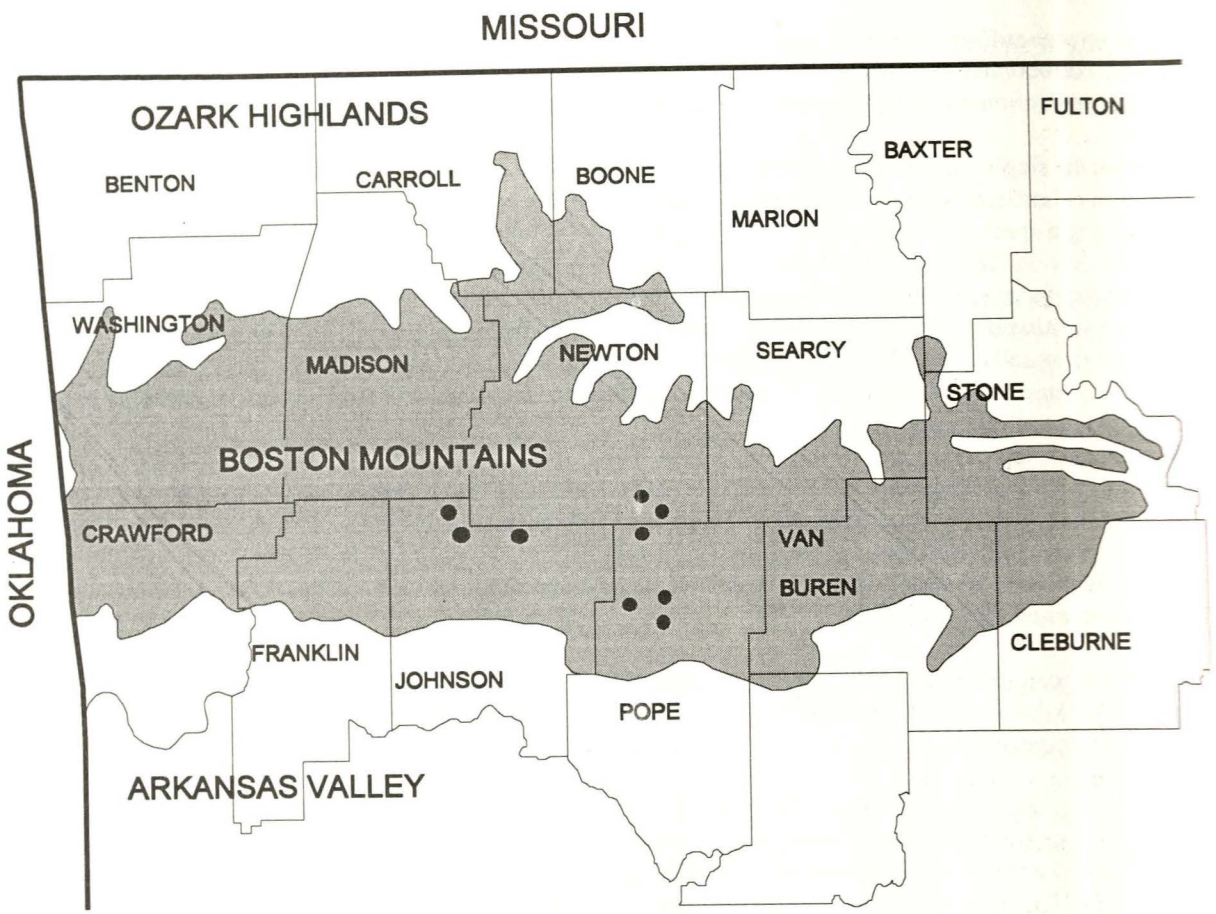


Figure 1. Location of study areas.

Soils on mountaintops and slopes usually have shallow to medium depth and are represented by medium-textured members of the Hartsells, Linker, and Enders series (Typic Hapludults). They are derived from sandstone or shale residuum, and their productivity is medium to low. In contrast, soils on mountain benches are deep, well-drained members of the Nella and Leesburg series (Typic Paleudults). They developed from sandstone and shale colluvium, and their productivity is medium to high. Rocks in the area are alternating horizontal beds of Pennsylvanian shales and sandstones.

Annual precipitation averages 46 to 48 in., and March, April, and May are the wettest months. Extended summer dry periods are common, and autumn is usually dry. The frost-free period is normally 180 to 200 days long.

Study Description

The regulation study design is a split-plot factorial layout, replicated three times, with aspect as the main plot treatment and residual stand density and regulation as sub-plot treatments. The aspects are north-east and south-west, residual densities are 65 and 85 $ft^2$/ac, and the regulation methods are free thinning and structural control. Although the study is a straightforward test of group selection with two residual stand treatments (i.e., free thinning versus structural control), it can also be used to evaluate single-tree selection by analyzing the structural control treatment.

Stand density treatments. The two density levels are 65 and 85 $ft^2$/ac of basal area in trees 5.6 in. and larger. The overstory density treatments were applied on the plot and buffer areas outside the group opening. The 65 $ft^2$ density level approximates the B-level of stocking for upland oak stands and should be near the optimum for stand and crop tree growth. The 85 $ft^2$ density represents about 70-75% stocking and is appropriate for a first thinning in older stands that have relatively high stocking levels. If competing subcanopy stems are effectively controlled, the higher residual stocking level might favor the development of oak reproduction and retard redevelopment of competing understory.

Regulation techniques. The regulation techniques applied to the residual stand are (1) area regulation with free thinning and (2) area regulation with structural control. Group selection will be used with area control. Assuming an 80-year conversion period and a 10-year cutting cycle, one-ninth of the area would be regenerated each cutting cycle.

Opening size is approximately 2.0 times the average height of the dominant trees in the adjoining stand. This will result in an opening size of about 0.4 ac in typical stand conditions. Selection of the initial group opening subplots were based on the following criteria: (1) presence of large reproduction of desirable species, (2) presence of saplings of desirable species, (3) sprouting potential of desirable species in the small poletimber class, and (4) overstory stocking. A subsequent group opening will be created every 10 years. Location of future openings will be based upon the presence of adequate advance reproduction.

In the first regulation method, the residual stand (the area not in group openings) was cut to the target density by free thinning. Trees were thinned in the following priority: (1) larger cull and defective trees, (2) competing trees of poor form and quality, and (3) intermediate and suppressed trees of lower quality and value. The primary objective was to improve residual stand quality and vigor.

In the second method, the residual stand was cut back to a target stand structure. The target stand structure has a minimum dbh of 5.6 in., a maximum dbh of 18.5 in., a $q$ of 1.3 (assuming 2-in. diameter classes), and residual densities of 65 and 85 $ft^2$/ac (Figure 2). Law and Lorimer (1989) suggested $q$ values of 1.3, 1.5, or 1.7 for 2-inch classes in upland hardwoods, and Smith and Lamson (1982) recommended a 1.3 $q$-value for sawtimber production and higher $q$'s for smaller-product objectives. Because our objective is quality sawtimber production, we chose a $q$ of 1.3. We selected the maximum dbh to produce a grade one butt log, also in accordance with a quality sawtimber objective. Although this target structure may be eventually created, current structure and species composition of our oak-hickory stands make the target structure impossible to attain with the first cut and could be silviculturally unsound if rigidly applied.

For example, most of our oak-hickory stands are essentially even-aged but are more irregular than typical even-aged stands regenerated from clearcuts. Our Boston Mountain stands tend to have an older-age component of 5-10 trees per acre that are generally larger than the main stand; these trees are often culls or very poor in quality. Moreover, we have many more stems in the small diameter classes (6-11 in. dbh) than are typical for even-aged stands. Many of these small stems are tolerant, noncommercial or low-value species that have increased in numbers and size as the overstory has matured. Moreover, the suppressed low-vigor stems of desirable species in these sizes have a low probability of responding to release. Therefore, rigid application of a target stand structure with no regard to the quality of the residual trees can diminish stand quality, a major offense to good silviculture.

When marking to achieve the residual structure for areas outside openings, we divided the trees into four size classes—small poletimber (6-8 in. dbh), large poletimber (9-11 in. dbh), small sawtimber (12-15 in. dbh), and large sawtimber (>15 in. dbh). We calculated the target residual basal area for each class and marked the stand to conform to the target. However, when marking in the poletimber classes, we discriminated against the noncommercial and low-value species and did not always achieve the target structure for the small poletimber class. In these cases, we left more stocking in better-quality trees in the larger classes. We did, however, leave some low vigor oak stems in the small diameter classes to evaluate survival and growth. The main goal was to leave the residual basal area in the best quality trees available on the plot.

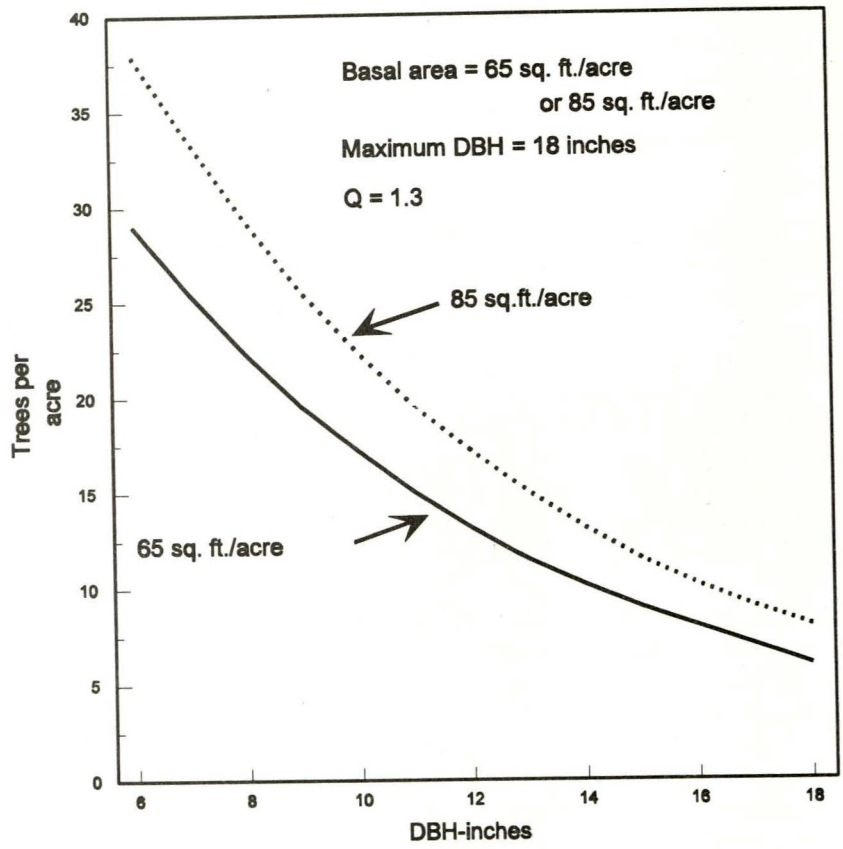


Figure 2. Target stand tables for plots regulated by structural control.

Plot location and layout. The study was installed on the Ozark National Forest in well-stocked hardwood sawtimber stands with no history of previous cutting for at least 50 years. Twenty-four 11-ac plots were installed in nine forest stands on the Buffalo, Bayou, and Pleasant Hill Ranger Districts (Figure 1). Study plots were located on north/east and south/west facing mountain slopes and benches and in oak-hickory stands that are representative of the sites and stand conditions that are designated for uneven-aged management by the Ozark National Forest.

Study plots were replicated by National Forest Districts with 8 plots established on each District according to the following schedule:

| District | Number of plots by aspect | | Preharvest measurements | Understory treatment | Harvest | Postharvest Measurement |
|---|---|---|---|---|---|---|
| | North | South | | | | |
| Buffalo | 4 | 4 | Winter 1992 | Summer 1992 | Summer 1993 | Fall 1993 |
| Bayou | 4 | 4 | Winter 1993 | Summer 1993 | Summer 1994 | Fall 1994 |
| Pleasant Hill | 4 | 4 | Winter 1994 | Summer 1994 | Summer 1995 | Fall 1995 |

Harvesting was accomplished by each National Forest District using standard timber sales. Logging at all study locations utilized chainsaw felling and tree-length skidding by standard rubber-tired skidders.

Sample plots consist of a 7.2-ac net plot plus a 66-ft buffer for a total of 11 ac (Figure 3). In addition, each 7.2-ac net plot is subdivided into 12 0.6-ac subplots. Of the 12 subplots on each plot, 3 subplots located on one end of the plot (numbers 1-3 or 10-12) were randomly selected for the competition study. The remaining 9 subplots were used in the regulation study.

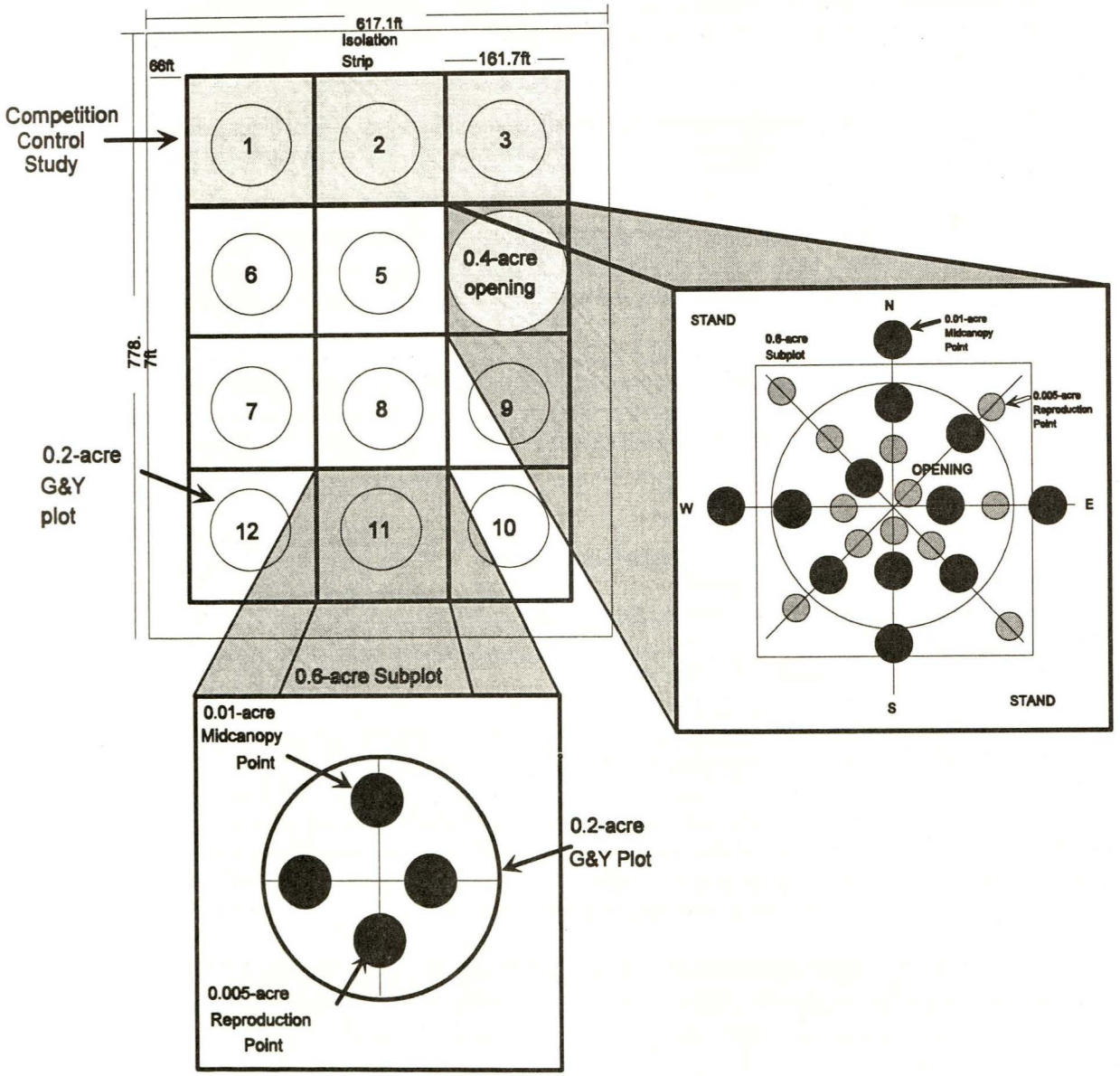


Figure 3. Layout of the plot, subplot, and sampling points for saplings and reproduction in the 0.2-ac growth plot and group opening.

Preharvest measurements. A complete tally of all overstory trees, dbh 5.6 in. and larger, was taken by species, tree class [per Grosenbaugh 1955 (see footnote, Table 3)], and 1-in. dbh classes. Saplings (0.6-5.5 in. dbh) were sampled by species and 1-in. dbh class on 48 0.01-ac sample points on each plot with 4 points permanently located within the center 0.2 ac of each 0.6-ac subplot (Figure 3). Preharvest reproduction (all stems smaller than 0.6 in. dbh) was sampled by species and 1-ft height class on a series of four 0.005-ac sample points on each subplot (Figure 3). In group opening subplots, all oak, ash, cherry, and walnut reproduction was identified and mapped on twenty-four 0.005-ac sample points (Figure 3). For these stems, height and basal diameter (1-in. above groundline) were measured to the nearest 0.1 ft and 0.1 in., respectively. The reproduction sample points are also permanently located and will serve to monitor reproduction development over time.

**Competition control.** Different treatments for controlling competing reproduction and midcanopy stems were applied to the residual stand and group opening:

> Residual stand—Unmerchantable stems of desirable species larger than 5.6 in. dbh were cut to promote sprouting. Stems of undesirable species taller than 4.5 ft were cut, and the stumps were treated with herbicide.
>
> Group opening—Unmerchantable stems of desirable species larger than 5.6 in. dbh were cut to promote sprouting. Stems of undesirable species taller than 1.0 ft were cut, and the stumps were treated with herbicide.

For this study, the species to be favored are: the hickories (*Carya* sp.), white ash (*Fraxinus americana* L.), black cherry (*Prunus serotina* Ehrh.), black walnut (*Juglans nigra* L.), black oak (*Quercus velutina* Lam.), northern red oak (*Q. rubra* L.), and white oak (*Q. alba* L.). Also, on poorer south and west aspect sites, shortleaf pine is considered desirable.

Postharvest measurement. Overstory growth and yield plus residual midcanopy stems were measured on a series of 0.2-ac circular plots located in the center of each 0.6-ac subplot (Figure 3). On each plot all overstory trees 5.6 in. dbh and greater were numbered and mapped by azimuth and distance from plot center. The following information was collected:

(1) diameter to nearest 0.1 in.,
(2) total height for a sample of trees in each 1-in. dbh class,
(3) log grade for all sawtimber trees,
(4) damage to crowns and boles resulting from logging,
(5) tree quality, and
(6) age of selected dominants or codominants.

Also, residual saplings (0.6-5.5 in. dbh) on four 0.01-ac sample points within each plot were mapped by azimuth and distance from point center and dbh measured to the 0.1 in. (Figure 3).

The influence of the group opening on the stem quality and growth of adjoining trees will be evaluated on selected sawtimber- and poletimber-sized trees bordering each opening. Eight dominant/codominant oaks and eight intermediate/overtopped oaks adjacent to the opening were selected. For each tree, species, crown class, dbh, butt log grade, and number of epicormic sprouts by 16-ft logs to 32 ft were recorded.

We will monitor the effects of the overstory and understory treatments on overstory growth and yield, subcanopy development, and reproduction survival and growth at 5-year intervals. The plots will be cut every 10 years.

## RESULTS AND DISCUSSION

### Preharvest Conditions

Overstory. The overstory basal area was remarkably uniform across all plots, varying from 92 to 114 ft²/ac for north aspects and from 91 to 112 ft²/ac for plots on the south aspects. The mean stand age and range for north aspects was 79 years and 71 to 93 years, respectively. Stands on south aspects were slightly younger with a mean of 74 years and a range from 68 to 81 years. Red or white oak site index on north aspects ranged from 62 to 72 ft (base age, 50 years) and 55 to 69 ft for south aspects. Mean site index was 67 ft and 62 ft for north and south aspects, respectively.

Total basal area was almost 100 ft²/ac for both aspects (Table 1). The basal area in desirable species is 90 percent for both aspects, and the oaks comprise about 80 percent of total basal area. There were some minor differences in species mix by aspect. Basal area for red oaks and ash-cherry-walnut was more on north aspects, while white oak basal area averaged more on south aspects.

Table 1. Preharvest species composition by basal area and aspect for the overstory (dbh ≥ 5.6 in.)

| Species group[1] | Aspect[2] | |
|---|---|---|
| | North | South |
| | - - - - - - - - Basal area (ft²/ac) - - - - - - - - | |
| Understory species | 1.5 | 2.9 |
| Other overstory species | 8.1 | 6.8 |
| Hickory-shortleaf pine | 8.2 | 6.5 |
| Ash-cherry-walnut | 2.2 | 1.4 |
| Red oaks | 43.2 | 34.6 |
| White oaks | 35.2 | 47.3 |
| All species | 98.4 | 99.5 |

[1] White oaks: White oak, post oak
Red oaks: Black oak, northern red oak
Ash-cherry-walnut: White ash, black cherry, black walnut
Hickory-shortleaf pine: Hickories, shortleaf pine
Other overstory species: Basswood, beech, blackgum, cucumber tree, sugar maple, sweetgum
Understory species: Blackhaw, black locust, buckeye, devil's walkingstick, dogwood, elm, hackberry, hophornbeam, Indian cherry, mulberry, Ozark chinkapin, papaw, persimmon, red cedar, redbud, red haw, sassafras, serviceberry, spice bush, tree huckleberry, umbrella magnolia, wild plum, witch hazel, miscellaneous understory

[2] Means are based on 12 7.2-ac plots for each aspect.

There were some differences in distribution of product classes by aspect (Table 2). There was more pulpwood basal area on south slopes, but the basal area for large sawtimber was greater on north slopes. These product distributions demonstrate that south aspects have more, smaller trees, while north aspects tend to have larger trees.

Table 2. Preharvest structure by product class and aspect for the overstory (dbh ≥ 5.6 in.)

| Product class | Aspect[1] | |
|---|---|---|
| | North | South |
| | - - - - - - - - Basal area ($ft^2$/ac) - - - - - - - - | |
| Small pulpwood (6-8 " dbh) | 12.7 | 16.8 |
| Large pulpwood (9-11" dbh) | 20.6 | 26.1 |
| Small sawtimber (12-15" dbh) | 32.0 | 31.2 |
| Large sawtimber (>15" dbh) | 33.1 | 25.4 |
| All classes | 98.4 | 99.5 |

[1] Means are based on 12 7.2-ac plots for each aspect

The tree class "grower" (Grosenbaugh 1955) designates trees that are the objective of management for quality timber. Table 3 shows that these crop trees are a much larger proportion of the stand on north aspects. This larger proportion occurs partly because the red oaks occur more often on north aspects, and red oaks tend to have a larger proportion of growers than white oaks. The incidence of culls and high-risk trees (riskers/killers/culls) occurs with equal proportions on both aspects.

Table 3. Preharvest structure by Grosenbaugh tree class and aspect for the overstory (dbh ≥ 5.6 in.).

| Grosenbaugh tree class[1] | Aspect[2] | |
|---|---|---|
| | North | South |
| | - - - - - - - - Basal area ($ft^2$/ac) - - - - - - - - | |
| Grower | 26.5 | 15.9 |
| Sleeper | 16.8 | 19.4 |
| Cipher/topper/slower | 45.4 | 53.3 |
| Risker/killer/cull | 9.7 | 10.9 |
| All classes | 98.4 | 99.5 |

[1] Adapted from Grosenbaugh (1955):

Grower: A merchantable tree that is vigorous and has no serious defects that would affect growth or desirability of the tree as potential sawtimber growing stock. A grower should also have the potential of developing a grade 1 butt log and have an expectancy of at least 0.90 of living until the next cutting cycle. Some people call such trees "crop trees" or "good growing stock."

Cipher: A merchantable tree whose expectancy of living for the next cutting cycle is at least 0.90, but does not meet the qualifications of a grower because of slow growth or undesirable characteristics, and is not competing with desirable reproduction or saplings. This tree can either be "financially mature" or may have limitations that disqualify it as a grower.

Sleeper: A cipher which has the potential to become a grower if it were released by removing competing trees.

Topper: A merchantable tree similar to a cipher but overtopping desirable reproduction or saplings.

Slower: The least potentially productive of several merchantable trees (but not riskers or killers—see below) competing in inadequate growing space. It should be cut in thinning.

Risker: A merchantable tree whose life expectancy for the next cutting cycle is less than 0.90. It should be cut to salvage potential loss through mortality.

Killer: A merchantable tree infested with contagious pathogens.

Cull: A tree that is merchantable size but not salable because of defect or other factors.

[2] Means are based upon 12 7.2-ac plots for each aspect.

Saplings. Total number of saplings did not vary much by aspect. This component was dominated by noncommercial understory species and less desirable overstory species (Table 4). Of the preferred species, white oak and hickory were most numerous. The hickories were most numerous in the smaller sapling classes, and white oaks had the most stems in the 3 to 5 in. dbh classes.

Table 4. Preharvest species composition for saplings (0.6-5.5 in. dbh) by aspect

| Species group[1] | Aspect[2] North | South |
|---|---|---|
| | Stems/ac | |
| Understory species | 658 | 492 |
| Other overstory species | 104 | 194 |
| Hickory-shortleaf pine | 39 | 16 |
| Ash-cherry-walnut | 19 | 19 |
| Red oaks | 5 | 7 |
| White oaks | 30 | 24 |
| All species | 855 | 752 |

[1] See footnote 1 in Table 1 for species list.
[2] Means are based on 12 7.2-ac plots for each aspect.

Reproduction. The oak reproduction is predominantly in the <1 ft height class (Table 5). Of rootstocks over 1 ft, the majority are less than 3 ft tall, and few are taller than 5 ft. Relatively few white oak reproduction rootstocks occur on either aspect. This is unusual because white oak reproduction is usually much more numerous than red oak, and it dominates south slopes. The hickory, ash, and cherry reproduction was much more numerous in the larger reproduction size classes. Noncommercial understory species dominated other species groups in number of rootstocks on both aspects; shortleaf pine was noticeably absent.

Table 5. Preharvest species composition for rootstocks (< 0.6 in. dbh)

| Species group[1] | Height class (ft) | Aspect[2] North | South |
|---|---|---|---|
| | | Stems/ac | |
| Understory species | <1.0 | 1,534 | 1,838 |
| | >1.0 | 2,433 | 1,866 |
| Other overstory species | <1.0 | 219 | 274 |
| | >1.0 | 365 | 446 |
| Hickory-shortleaf pine | <1.0 | 510 | 282 |
| | >1.0 | 443 | 259 |
| Ash-cherry-walnut | <1.0 | 259 | 210 |
| | >1.0 | 477 | 470 |
| Red oaks | <1.0 | 885 | 711 |
| | >1.0 | 92 | 145 |
| White oaks | <1.0 | 265 | 352 |
| | >1.0 | 33 | 43 |
| | | Stocking Value[3] | |
| All oaks | | 2 | 3 |

[1] See footnote 1 in Table 1 for species list.
[2] Means are based on 12 7.2-ac plots for each aspect.
[3] Stocking values computed from Sander and others (1984), based on 56 1/735-ac sample points per 7.2-ac net plot.

Oak stocking value (Sander and others 1984) is the contribution of reproduction to stand stocking at age 20 and is based upon reproduction size (height and ground diameter), aspect, and slope position. It is a method to determine before final harvest whether or not advance reproduction can adequately regenerate an oak stand. A stocking value of 30 or more indicates adequate oak reproduction. Based upon samples of 56 1/735-ac points per plot, the average stocking value was 2 and 3 for north and south aspect plots, respectively (Table 5). These low values indicate that oak reproduction is inadequate, and treatments are required to favor the establishment and growth of additional oaks.

Although oak stocking in the initial group openings will be low, large ash and cherry reproduction is present on more mesic sites and will supplement the oak reproduction developing from advance stems and stump sprouts. On poorer sites, existing hickory saplings and oak stump sprouts will supplement existing oak reproduction. Future openings will be installed in those subplots with adequate advance oak reproduction.

Postharvest Conditions

Overstory. The treatments affected residual species composition. The more desirable species were retained, and the other groups were discriminated against when making the cut. Therefore, there was a large reduction in the understory species and other overstory species groups (Table 6). We now have from 92 to 99% of the residual basal area in desirable species. The 85 $ft^2$ basal area treatments for both structural control and free thinning did not give as much freedom in molding species composition as the 65 $ft^2$ treatment, because not as much basal area was removed. There was no apparent difference in species composition between the two cutting techniques.

Table 6. Postharvest species composition by aspect and treatments for the overstory (dbh ≥ 5.6 in.)

| | Residual basal area ($ft^2$/ac) | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | North aspect | | | | South aspect | | | |
| | 65 | | 85 | | 65 | | 85 | |
| Species group[1] | Free thinning[2] | Structural control | Free thinning | Structural control | Free thinning | Structural control | Free thinning | Structural control |
| | $ft^2$/ac | | | | | | | |
| Understory species | 0.1 | 0.4 | 1.3 | 0.9 | 0.6 | 0.6 | 1.2 | 1.8 |
| Other overstory species | 0.5 | 0.9 | 2.6 | 4.7 | 0.8 | 1.6 | 6.1 | 3.7 |
| Hickory-shortleaf pine | 4.1 | 4.4 | 8.3 | 9.6 | 4.4 | 4.0 | 5.2 | 5.8 |
| Ash-cherry-walnut | 0.5 | 1.0 | 1.8 | 2.9 | 0.5 | 0.6 | 0.4 | 2.1 |
| Red oaks | 27.4 | 31.5 | 35.9 | 39.4 | 24.9 | 19.6 | 23.2 | 33.5 |
| White oaks | 32.4 | 27.0 | 32.4 | 25.9 | 34.2 | 38.6 | 47.8 | 36.8 |
| All species | 65.0 | 65.2 | 82.3 | 83.4 | 65.4 | 65.0 | 83.9 | 83.7 |

[1] See footnote 1 in Table1 for species list.
[2] Means are based on 3 7.2-ac plots for each aspect-density-regulation combination.

The treatments reduced basal area from almost 100 $ft^2$/ac to 65 and 85 $ft^2$/ac, and most of the reduction was in sawtimber (Table 7). The heavier reduction in the large sawtimber product class occurred because it contained more culls and lower-quality trees. A major objective was to improve residual stand quality; therefore, culls and lower quality trees had the highest priority for removal regardless of stem size. As illustrated in Table 8, the largest reduction in basal area did occur in culls and the lower quality classes. Culls were reduced from 9 to 14% in preharvest conditions to 0 to 2% in the after-cut stands. Any culls that were left occurred in the 85 $ft^2$/ac treatment to meet residual basal area targets. The reduction in basal area was least in the grower and sleeper categories, which are the best potential crop trees. The proportion of basal area in these trees was increased.

Table 7. Postharvest product class distribution by aspect and treatments for the overstory (dbh ≥ 5.6 in.)

| Product size | Residual basal area ($ft^2$/ac) | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | North aspect | | | | South aspect | | | |
| | 65 | | 85 | | 65 | | 85 | |
| | Free thinning[1] | Structural control | Free thinning | Structural control | Free thinning | Structural control | Free thinning | Structural control |
| | $ft^2$/ac | | | | | | | |
| Small pulpwood (6-8" dbh) | 8.9 | 10.9 | 13.2 | 9.0 | 13.0 | 13.4 | 13.7 | 14.2 |
| Large pulpwood (9-11" dbh) | 18.1 | 16.7 | 19.3 | 15.4 | 21.3 | 19.4 | 22.0 | 24.1 |
| Small sawtimber (12-15" dbh) | 23.2 | 22.5 | 25.8 | 30.5 | 20.6 | 22.6 | 28.2 | 26.0 |
| Large sawtimber (>15 dbh) | 14.8 | 15.0 | 24.0 | 28.5 | 10.5 | 9.6 | 20.0 | 19.4 |
| All product sizes | 65.0 | 65.1 | 82.3 | 83.4 | 65.4 | 65.0 | 83.9 | 83.7 |

[1] Means are based on 3 7.2-ac plots for each aspect-density-regulation combination.

Table 8. Postharvest Grosenbaugh tree class distribution by aspect and treatments for the overstory (dbh ≥ 5.6 in.)

| Grosenbaugh tree class[1] | Residual basal area ($ft^2$/ac) | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | North aspect | | | | South aspect | | | |
| | 65 | | 85 | | 65 | | 85 | |
| | Free thinning[2] | Structural control | Free thinning | Structural control | Free thinning | Structural control | Free thinning | Structural control |
| | $ft^2$/ac | | | | | | | |
| Grower | 22.7 | 26.0 | 24.9 | 29.7 | 15.3 | 13.8 | 18.2 | 14.5 |
| Sleeper | 15.8 | 18.2 | 17.8 | 15.6 | 19.7 | 17.9 | 17.4 | 22.5 |
| Cipher/topper/slower | 26.6 | 20.9 | 38.3 | 38.1 | 30.4 | 33.3 | 46.3 | 45.2 |
| Risker/killer/cull | 0.0 | 0.0 | 1.3 | 0.0 | 0.0 | 0.0 | 2.0 | 1.5 |
| All classes | 65.0 | 65.1 | 82.3 | 83.4 | 65.4 | 65.0 | 83.9 | 83.7 |

[1] See footnote 1 in Table 3.
[2] Means are based on 3 7.2-ac plots for each aspect-density-regulation combination.

Figure 4 shows before- and after-cut stand structures by treatments. There is little apparent difference between the free thinning and structural control treatments after the initial cut for either density class. This lack of difference is the result of removing lower quality trees to improve stand quality irrespective of cutting method or residual density. Figure 4 also shows that there is a deficit of small diameter trees for both residual basal areas and aspects when compared to the target stand tables for the structural control treatments. This situation is further complicated by the presence of many smaller noncommercial or lower vigor stems that have a poor likelihood of responding to release. Many are ciphers. In marking these stands, higher priority was given to removing culls, high risk trees, trees competing with higher quality trees, and noncommercial species. Lower priority was given (1) to removing smaller, low-vigor trees that do not compete with better quality stems or (2) to attaining structure. In marking the structure plots, some low-vigor oaks in the intermediate and overtopped crown classes were also left to evaluate their response to release.

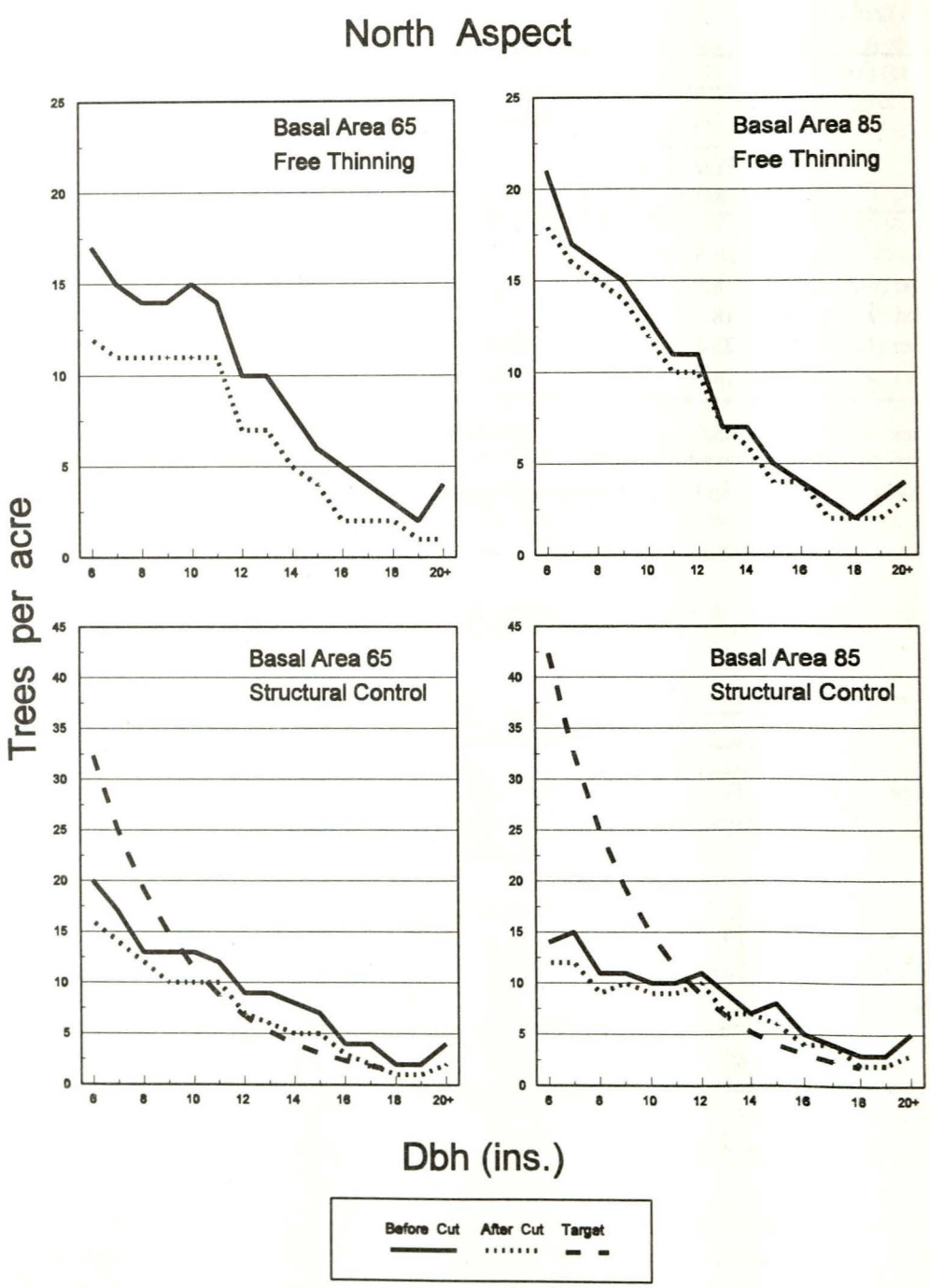


Figure 4a. Before-cut, after-cut, and target stand tables (structural control plots only) for north aspect plots by residual basal area and type of thinning.

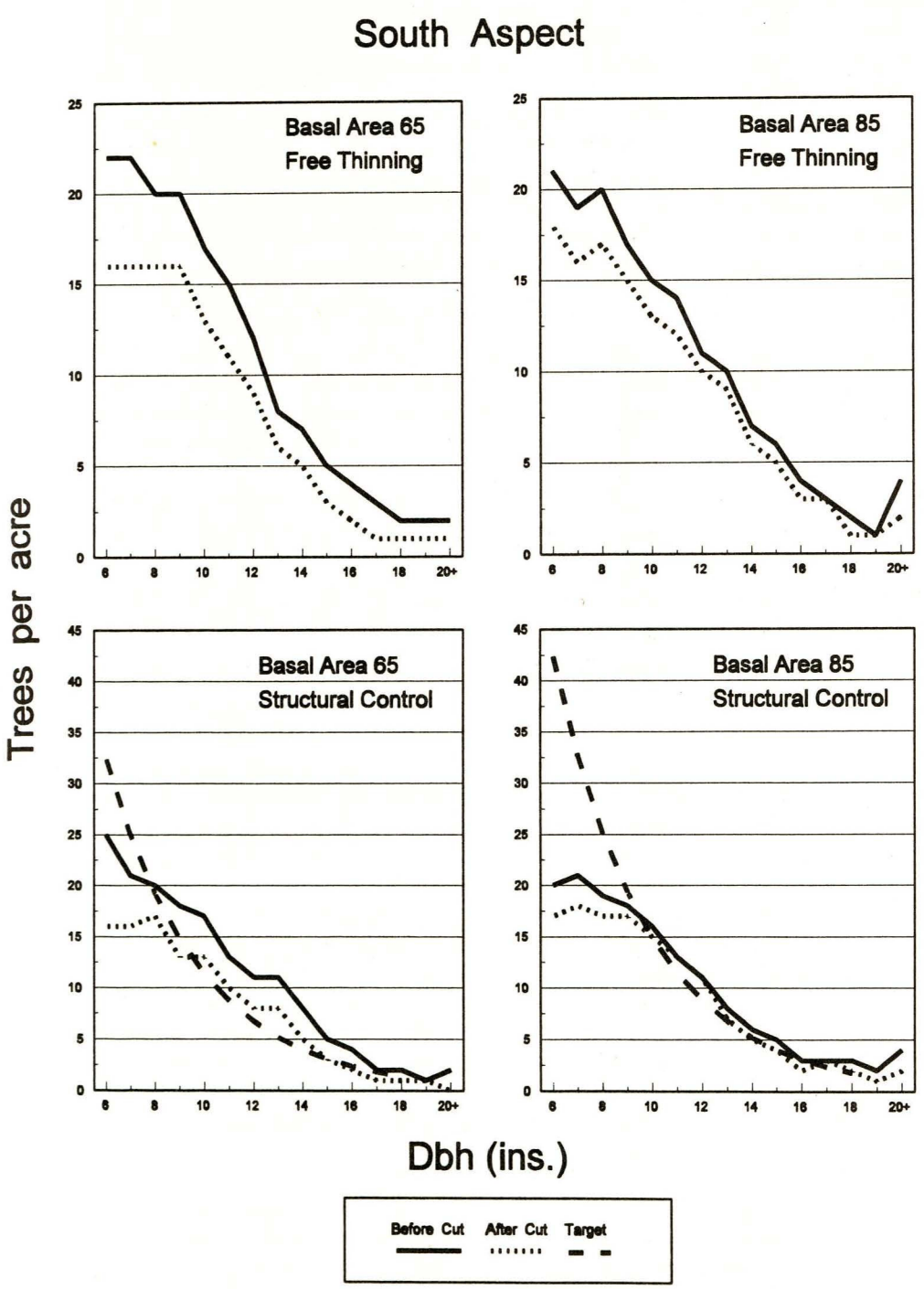


Figure 4b. Before-cut, after-cut, and target stand tables (stuctural control plots only) for south aspect plots by residual basal area and type of thinning.

Saplings. In addition to the deficiency in small overstory stems, there are also few residual saplings of desired species on both aspects (Table 9). Although sapling stocking standards have not been developed for uneven-aged oak-hickory stands, we would expect it to exceed at least 100 stems per acre of desired species. The number of red oak saplings is very low, while the number for white oak is somewhat greater, possibly indicating the difference in the shade tolerances between these species.

Table 9. Postharvest sapling (0.6-5.5 in. dbh) inventory by species group and aspect

| Species group[1] | Aspect[2] | |
|---|---|---|
| | North | South |
| | - - - - - - - - - - Stems/ac - - - - - - - - - | |
| Understory species | 10 | 8 |
| Other overstory species | 1 | 3 |
| Hickory-shortleaf pine | 19 | 11 |
| Ash-cherry-walnut | 10 | 14 |
| Red oaks | 3 | 3 |
| White oaks | 17 | 15 |
| All species | 60 | 54 |

[1] See footnote 1 in Table1 for species list.
[2] Means are based on 12 7.2-ac plots for each aspect.

Reproduction. Reproduction was not inventoried after treatment. Competing stems taller than 4.5 ft were cut and treated with herbicide on the regulation study plots. In group openings, competing stems taller than 1 ft were cut and treated with herbicide. These treatments have been shown to be effective in increasing growth of established oak reproduction on upland sites in the Boston Mountains of Arkansas (Graney and Murphy 1995), the Missouri Ozarks (Schlesinger and others 1993), and southern Appalachians (Loftis 1990). In the Boston Mountains, other desirable species—such as black cherry and white ash—also respond significantly to competition control (Graney and Murphy 1995). While we know that some control of competing stems is required, the question is how much and how does it vary with site quality or aspect.

## CONCLUSIONS

Though this study has just begun, experience gained in study installation and analysis of stand conditions before and after harvest leads to some tentative conclusions. They are:

1. Oak and other desirable reproduction on mesic and dry mesic Boston Mountain sites may be inadequate for regeneration cuts. A tolerant understory on these more mesic sites has prevented the development of adequate oak reproduction. The establishment and development of desirable reproduction will be the driving force for successful application of either uneven-aged cutting method. Therefore, understory competition control treatments are essential to promote establishment and growth of oak and other desirable reproduction. Experience with uneven-aged management of the shade-intolerant southern pines indicates that intensive competition control treatments may be required, especially on the more mesic sites.

2. Successful application of single-tree selection requires the eventual development of a well balanced stand structure. However, the present stands have a deficiency of desirable reproduction, sapling, and small poletimber trees. Filling these deficits in many cases will depend upon the establishment of desirable reproduction and its subsequent recruitment into these deficient size classes. Existing sapling and poletimber trees are of

the same age class as the overstory. They have been overtopped for a long time and generally have small crowns and low vigor. Therefore, these stems may not respond to release and possibly cannot be counted on to grow into the larger sizes as quality sawtimber. Until the development of a desirable stand structure, there will probably be no real difference between structural control and free thinning.

3. The stands before treatment had a high proportion of merchantable stems in the grower and sleeper categories, 44 and 35% of the total basal area for north and south aspects, respectively. These proportions were increased by the initial cut. Therefore, these stands have a strong potential for producing quality sawtimber, especially if residual stand quality is continually improved by subsequent cutting. Of course, the potential is much greater on the more mesic north slopes where better growth produces larger trees. The emerging market for hardwood chips in this region can give additional impetus to upgrading stand quality by providing an outlet for low-quality small material.

4. Recognizing quality is necessary for successfully applying either method to stands similar to the ones in this study. Blind, rigid application of structural control to these stands will degrade stand quality. It will be easier, perhaps, to achieve stand quality objectives using free thinning. Free thinning allows more flexibility in marking the stand because there are no stand structure constraints.

5. Residual density should be chosen with care. If the density is too high, the cut may not be operable, may not promote the growth of the residual trees, and may not afford the opportunity to upgrade stand quality. For example, the 85 $ft^2$/ac treatment was not operable and resulted in removing only a portion of the culls and high risk trees. If the residual density is too low, stand growth can be reduced, stem quality may be degraded from increased epicormic sprouting, and rapid redevelopment of competing understory may occur.

6. Although the oaks have been traditionally favored in management in this region, a shift in species composition may be occurring that might cause forest managers to reevaluate this favoritism in some situations. This study and others (Graney 1989, Graney and Murphy 1995, Graney and Rogerson 1985) have noted the presence of large advance reproduction of cherry and ash on more mesic sites. These are also favored species in management, though their numbers will probably not dominate the stand as the oaks do now. The presence of these species may give forest managers more options, though cherry does not attain the quality and size associated with this species in other regions.

Despite the increased interest in using uneven-aged silviculture in upland oak stands in the Boston Mountains and elsewhere, we have neither a reservoir of experience in using these techniques with upland oaks nor are there studies that have matured enough to yield guidelines for management. Moreover, some fragmentary evidence indicates that single-tree selection can be used in more dry and xeric conditions with success, which contradicts accepted wisdom accumulated in more mesic situations. Hopefully, as this study and others accumulate results, the decision to use uneven-aged silviculture in this forest type will be based more on sound facts and owner objectives instead of aversion to other cutting methods and the reigning political climate.

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# DIAMETER GROWTH OF TREES IN AN UNEVEN-AGED OAK FOREST IN THE MISSOURI OZARKS

Edward F. Loewenstein[1], Paul S. Johnson[1], and Harold E. Garrett[2]

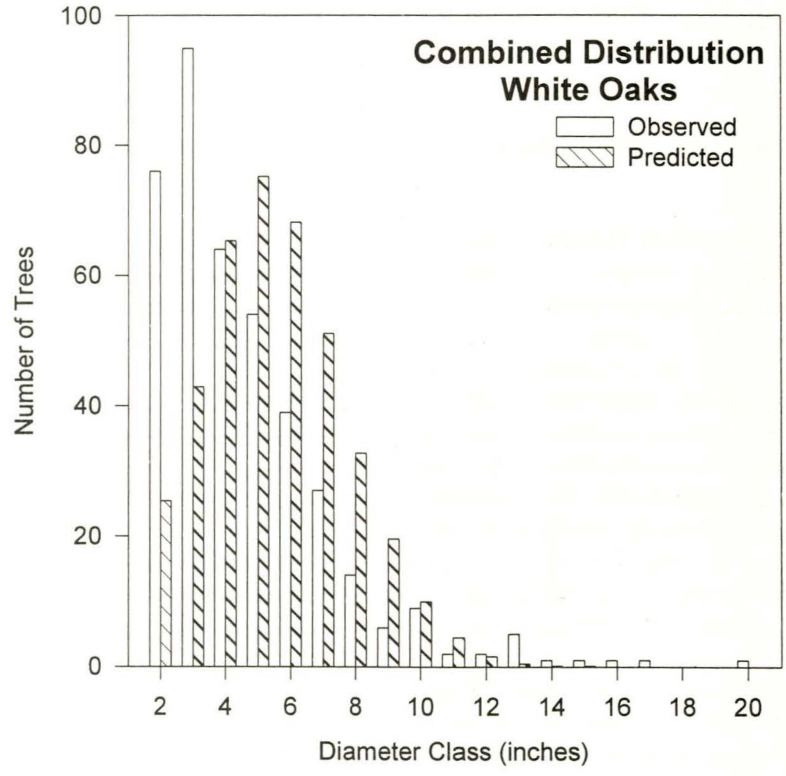
Abstract: We tested the efficacy of even-aged stand tables for predicting diameter growth of trees in uneven-aged oak stands. The study was based on the age- and diameter-structure of the Pioneer Forest, a 156,000-acre, privately owned oak forest in the Ozark Highlands of Missouri. The forest has been managed by single-tree selection since 1954.

Observed diameter distributions were drawn from a random sample of 60 oaks ≥1.6-inches dbh on each of 10 1-acre plots. The plots were randomly located across one square mile of the forest. Dbh and the age of each tree at a height of 1 foot above groundline were determined. We compared the observed diameter distributions with those predicted by Schnur's even-aged stand tables for upland oaks. Predicted diameter distributions were derived by aggregating the age distribution of the observed sample into 10-year age classes and projecting the growth of each age class based on the stand tables. The resulting even-aged distributions were then combined to form a predicted uneven-aged diameter distribution for a single stand. Chi-square and Kolmogorov-Smirnov (KS) one-sample tests were used to test the null hypothesis of no difference between observed and predicted distributions.

Separate comparisons were made for red oak and white oak species groups. Statistical tests were applied to the composite distribution and to each 1-acre plot. Results were the same for chi-square and KS tests. In general, even-aged stand tables did not accurately describe the diameter growth of individual cohorts within an uneven-age stand. The range of observed diameters was greater in the uneven-aged stand than would be expected had the trees grown in even-aged stands at 100% ("normal") stocking. Observed composite distributions for the red oak (n=180) and white oak (n=398) species groups differed from the predicted distribution ($p<0.001$). Of the 20 comparisons at a scale of one-acre, 15 did not differ significantly. However, at the smaller scale, the paucity of statistically different comparisons may have been the result of the reduced power of tests associated with small populations.
The lack of agreement between observed and predicted distributions may largely be related to differences in growth and mortality of trees on the Pioneer Forest and those expected in normally stocked even-aged stands. Because the Pioneer Forest is managed at a relatively low density (average stocking is 67% for trees ≥1.6 inches dbh), we might expect density-dependent mortality rates of trees to be relatively low and growth rates to be relatively high. In turn, this may explain the larger-than-expected number of small trees and the extended distribution of large trees in the observed distributions. The discrepancy between observed and predicted was greatest for small-diameter white oaks. The substantially greater-than-expected numbers of white oaks in the 2- and 3-inch dbh classes emphasizes the inadequacy of the even-aged stand tables to express the shift to a negative exponential diameter distribution at reduced stand density (Figure 1).

[1] Silviculturist and Principal Silviculturist, respectively, USDA Forest Service, North Central Forest Experiment Station, 1-26 Agriculture Building, Columbia, MO 65211.

[2] Professor, School of Natural Resources, University of Missouri, 1-31 Agriculture Building, Columbia, MO 65211.



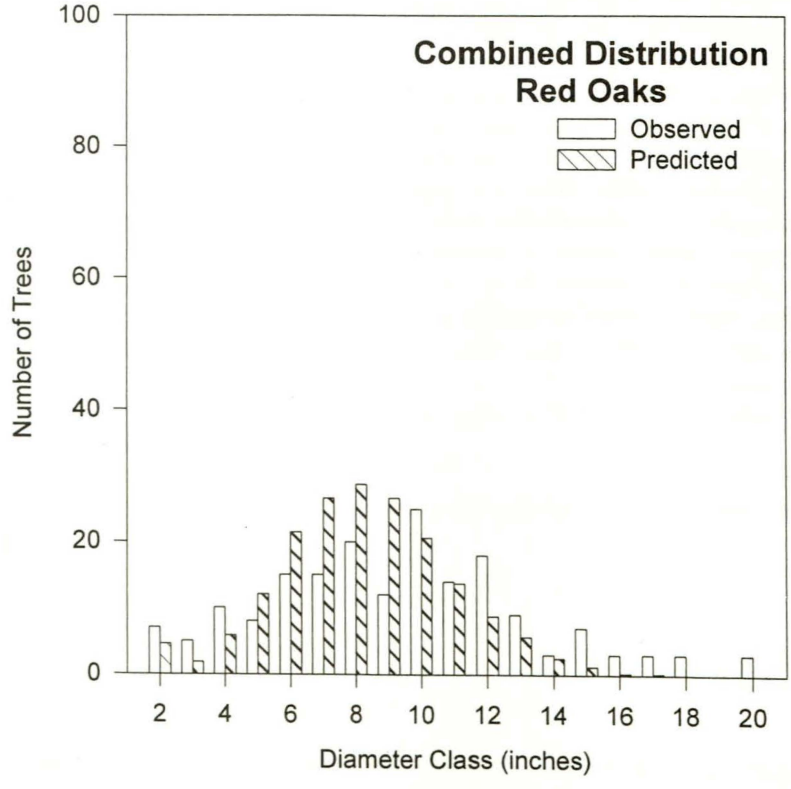




Figure 1. Observed and predicted diameter distributions by oak species group.

# STABILITY OF DIAMETER DISTRIBUTIONS

# IN A MANAGED UNEVEN-AGED OAK FOREST IN THE OZARK HIGHLANDS

Zhiming Wang,[1] Paul S. Johnson,[2] H. E. Garrett,[1] and Stephen R. Shifley[2]

Abstract: We studied a privately owned 156,00-acre oak-dominated forest in the Ozark Highlands of southern Missouri. The forest has been managed by the single-tree selection method since 1952. Using 40 years of continuous forest inventory records, we analyzed the stability of the shape of tree diameter distributions at the forest-wide scale. Results show that for trees ≥5-inches dbh, diameter frequency distributions closely conform to the negative exponential function for the three species groups and three stand density classes compared. Values of the slope coefficient for the negative exponential model, and thus the ratio of number of trees in successive dbh classes (*q*), were stable at the highest stand density class (>57% stocking) but tended to be unstable at lower stocking levels (<40% and 40-57%) for the three species groups considered (all species, red oaks, and white oaks). Among species groups, white oaks were least stable and the all-species group most stable. For any given 15- or 30-year interval, *q* (for 1-inch dbh classes) ranged from 1.18 to 1.44 (1.4 to 2.1 for 2-inch classes). When trees from 1.6 to 4.9 inches were included, the power function fit the observed distributions better than the negative exponential function. However, when observed distributions were fit separately to the range of diameters above and below 9 inches (near the minimum cutting diameter), the negative exponential model in most cases fit both segments of the distribution better than the power function. Statistical tests of the equality of the two parameters of the negative exponential model indicated there is significant discontinuity between dbh distributions above and below the minimum cutting diameter. The overall instability of diameter distributions appears to be related to self-limiting oscillations in slope shape and discontinuities that, within a cutting cycle, pivot or "seesaw" about the minimum cutting dbh of 10 inches. Such instability would not appear to adversely affect sustaining a negative exponential diameter distribution and thus the silvicultural system.

[1] University of Missouri, School of Natural Resources, Columbia, MO 65211.

[2] USDA Forest Service, North Central Forest Experiment Station, Columbia, MO 65211.

# FOREST INGROWTH PREDICTION MODEL FOR THE NORTHEASTERN UNITED STATES

Linda S. Gribko[1]

Abstract: In the last 20 years, there has been a revival of interest in the use of uneven-aged forest management techniques in the production of timber and forest amenity values. Uneven-aged management is coming into renewed favor especially among non-industrial private landowners in the northeastern United States. The practice allows periodic timber removals on relatively small tracts and, in addition, has been suggested as an aesthetic alternative to clearcutting.

Most of the forest growth and yield simulators available for use in the Northeast were developed with even-aged silvicultural systems in mind; therefore, they make little or no provision for the ingrowth of trees into the smallest inventoried diameter class. In addition, the few ingrowth models that have been developed have largely ignored species composition due to the complexity of the eastern deciduous forest types and the modeling effort required. Existing simulators are consequently of very limited value in the prediction of the growth and yield of stands managed by uneven-aged selection methods.

This paper describes a new forest ingrowth model developed for the northeastern United States. The model has been calibrated using data from 3,159 USDA FS Forest Inventory and Analysis (FIA) plots located in 14 northeastern states. The data set includes measurements of all trees over a 5" diameter threshold and, since the plots were remeasured on a 7-year cycle, a record of ingrowth between measurements.

The model was developed in two stages. First, a non-linear equation was developed to predict ingrowth trees per acre. Independent variables included number of trees per acre, average diameter of the trees, and a generalized stocking percent. Multivariate logistic modeling then was used to determine the species composition of the in-growth. A unique approach, based on individual species' tolerance of shading, was used in simultaneously fitting a set of logistic models to the data. The models have been validated using independent data sets from a wide range of northeastern forest types.

[1] Assistant Professor of Forest Management, West Virginia University, Division of Forestry, P.O. Box 6125, Morgantown, WV 26505.

# PLANTING OAKS IN THE CENTRAL HARDWOOD REGION:

# A SHELTERWOOD APPROACH

Dale R. Weigel and Paul S. Johnson[1]

Abstract: The success of oak seedlings planted under shelterwoods depends on where they are planted, initial seedling size, nursery undercutting treatments, and whether they are top-clipped before planting. These conclusions are based on planting northern red oak (*Quercus rubra* L.) and white oak (*Q. alba* L.) under shelterwoods in Indiana and Missouri. Three years after planting, the shelterwoods were completely removed. Results are based on planted tree growth and survival 5 to 7 years after shelterwood removal. For both species, planted tree survival and the probability of attaining dominance increased as initial basal diameter of seedlings increased. For a given initial seedling size, survival and dominance probabilities were lowest for seedlings that were neither undercut in the nursery nor top-clipped 8 inches above the root collar. Regardless of undercutting treatment, top-clipped seedlings always produced higher dominance probabilities than unclipped seedlings. For a given treatment, dominance probabilities were higher in Missouri than in Indiana because of the presence and rapid growth of yellow-poplar on the Indiana sites. Planting prescriptions designed to meet a specific oak stocking goal therefore should consider the expected competition after final harvest as well as the expected growth and survival of planted trees.

[1]Forester, North Central Forest Experiment Station, Bedford, IN; and Principal Silviculturist, North Central Forest Experiment Station, Columbia, MO, Forestry Sciences Laboratory, 1-26 Agriculture Building, University of Missouri, Columbia, MO 65211.

# GROUND COVER IN OLD-GROWTH FORESTS

# OF THE CENTRAL HARDWOOD REGION

Martin A. Spetich[1], Stephen R. Shifley[2], George R. Parker[3], and Felix Ponder, Jr.[4]

Abstract: Differences in ground cover (percent cover of litter, percent cover of vegetation and litter weight) in old-growth forests across this region are not well understood. We initiated a long-term study in a three-state region to enhance knowledge in this area. We present baseline results for ground cover and compare these data across productivity regions. Thirty 0.25-ac (0.1-ha) main plots and 120 subplots 27 $ft^2$ (2.5 $m^2$) in size were established in each old-growth forest and used to inventory ground cover. We estimated percent cover of: bare ground, boles of woody vegetation, dead leaves, downed woody material, exposed rock, and herbaceous cover. In addition, 240 subplots 1.35 $ft^2$ (0.125 $m^2$) in size were used to estimate dry weight of litter down to the mineral soil horizon. Trends among sites were compared to a regional productivity gradient. Across all sites, mean values for percent ground cover were: bare ground, 6% (SD 10%); boles of woody vegetation ≥ 3.3 ft tall, 2% (SD 1%); dead leaves and dead herbaceous vegetation, 85% (SD 10%); downed woody material, 5% (SD 1%); exposed rock, 1% (SD 2%); other, 1% (SD 2%). Mean litter weight across all sites was 6.8 tons/ac (SD 3.1 tons/ac) [15.3 metric tons/ha (SD 7 metric tons/ha)]. Differences in litter weight were not significant between northeast and southwest slopes. The inverse relationship of percent cover of exposed rock to regional productivity class was highly significant ($R^2$ = 0.88 for negative exponential model, p-value < 0.01). The positive linear relationship between litter weight and productivity class was also significant ($R^2$ = 0.55, p-value = 0.01).

## INTRODUCTION

Parker (1989) defines mesic old-growth in the central hardwood region as those forests with overstory canopy trees > 150 years old, little human-caused understory disturbance during the past 80 to 100 years, all-aged structure, multilayered canopies with dominant canopy trees from 30 to 65 in dbh and understories of late-seral, shade-tolerant trees. He also describes a mosaic of all-aged canopy gaps and significant standing and downed dead trees. Martin (1992) defines mixed mesophytic old-growth forests of the Appalachians similarly but with some additional characteristics such as canopy trees > 200 years old, canopy trees at 100 trees/ac, basal areas averaging 110 $ft^2$/ac and undisturbed soils with macropores. However, little work has been done on soils of old-growth forests of the region (Lindsey and others 1965, Auten 1941). Although total litterfall tends to increase with above-ground net primary production (O'Neill and DeAngelis 1981), this relationship of litterfall vs. productivity has not been tested for midwestern old-growth forests.

We established this study to collect baseline data as part of the initial stage of a long-term study of midwestern old-growth forests. Because litter weight was the most costly measurement in both time and money, we also explore some relationships between litter dry weight and ground cover that could help us increase efficiency of future inventories. Study objectives were to determine: (1) regional baseline statistics for percent cover and litter weights in midwestern old-growth forests as well as relative variability of these factors, (2) how these factors vary across a regional forest productivity gradient, and (3) the relationship between percent cover of dead vegetation and litter weight.

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[1,3] Postdoctoral Research Scientist and Professor, respectively, Purdue University West Lafayette, IN.
[2,4] Research Forester and Project Leader, respectively, USDA Forest Service, North Central Forest Experiment Station, USDA Forest Service, [2]Columbia, MO and [4]Jefferson City, MO.

## METHODS

### Site Selection

Old-growth forests were considered as those forests having significant amounts of deadwood, little human disturbance, large old trees, multilayered canopies and trees > 150 years old (Parker 1989). We selected 10 old-growth sites (Table 1) out of the public trust old-growth forests considered to be the least disturbed sites in Indiana, Illinois and Missouri.

Table 1.—Ten old-growth sites inventoried for percent ground cover and litter weight.

| STATE | OLD-GROWTH SITE | SITE CODE | PRODUCTIVITY REGION [a] | PRODUCTIVITY VOLUME [b] ($ft^3$/ac/yr) | DATES LITTER SAMPLED | STAND $ft^2$/ac | ANNUAL TEMPERATURE (degrees F) | ANNUAL PRECIPITATION (in) |
|---|---|---|---|---|---|---|---|---|
| INDIANA | LAUGHERY BLUFF | LA | UPLAND FLATS | 87 | JUNE 1993 | 91.4 | 51.6 | 41.9 |
| INDIANA | HEMMER WOODS | HE | LOWER WABASH | 87 | MAY 1993 | 109.1 | 55.8 | 43.1 |
| INDIANA | DAVIS WOODS | DA | NORTHERN | 86 | MAY 1992 | 139.9 | 49.3 | 36.6 |
| INDIANA | DONALDSON'S WOODS | DO | KNOBS | 86 | JULY 1992 | 140.6 | 53.4 | 47.3 |
| INDIANA | PIONEER MOTHER RNA | PM | KNOBS | 86 | JUNE 1992 | 123.2 | 53.4 | 47.3 |
| ILLINOIS | SPITTLER WOODS | SW | PRAIRIE | 77 | JUNE 1993 | 132.4 | 52.8 | 40.2 |
| MISSOURI | DARK HOLLOW | DH | PRAIRIE | 60 | MAY 1993 | 108.2 | 51.7 | 36.4 |
| MISSOURI | ENGELMAN WOODS | EW | RIVERBORDER | 56 | MAY-JUNE 1993 | 105.3 | 54.8 | 37.9 |
| MISSOURI | BIG SPRINGS | BS | EAST OZARKS | 54 | SEPT-OCT. 1992 | 104.8 | 54.9 | 47.0 |
| MISSOURI | ROARING RIVER | RR | SW OZARKS | 52 | JUNE-AUG. 1992 | 108.0 | 55.1 | 44.4 |

[a] Productivity region refers to the multi-county inventory regions used to report forest inventory statistics.

[b] Productivity volume is an estimate of the inherent capacity of forests to grow crops of industrial wood averaged across all upland oak forests in the productivity region

### Percent Cover

Thirty 0.25-ac (0.1-ha) main plots were systematically established to inventory forest overstory characteristics at each old-growth site. Main plot centers were at least 40 m apart within each forest. Within each plot, four subplots of 27 $ft^2$ (2.5 $m^2$) each were established to inventory percent ground cover of: bare ground, boles of woody vegetation (trees and shrubs > 3.3 ft [1 m] tall), dead leaves, downed woody material, and exposed rock. Values for these variables totaled 100%. Estimates for each subplot were made using a square frame with 10 equal intervals delineated on each side. Cover of live herbaceous and woody vegetation less than 3.3 ft (1 m) tall was also estimated. Values were averaged across all plots per site to obtain the mean site values that were used in subsequent analysis.

### Forest Litter

Litter samples were taken at eight subplots distributed across each main plot. At each subplot a square wooden template was used to delineate a 1.35 $ft^2$ (0.125 $m^2$) forest floor subplot. Forest floor litter was removed down to the mineral soil layer at each subplot. The eight subsamples were combined for a plot estimate of moist weight. Combined litter samples were either returned to the lab in their entirety for dry weight determination or a moist weight was taken in the field with a sample of litter returned to the lab for determination of a dry to moist weight ratio. The ratio of dry/moist weight of this sample was used to estimate the dry weight of the eight combined litter subsamples.

### Productivity Regions

There is a general gradient of increasing site productivity across the study area from southwestern Missouri to eastern Indiana. Direct estimation of site quality at a given site and direct comparison of site quality among sites is difficult. Due to the age of overstory dominants and the periods of suppression often associated with trees at these sites, measurement of site index is problematic. However, statewide inventories show clear patterns of increasing cubic foot volume productivity in upland-oak forests along this gradient. Data from statewide forest inventories were used to document the general gradient of increasing cubic foot volume productivity for upland oak forests across the three-state region (Hahn 1985, Hahn 1989, Smith and Golitz 1986, Brand and Walkowiak 1990). A regional productivity estimate was assigned to each old-growth tract based on the mean productivity for upland oak forests in the multi-county Forest Service inventory unit in which it falls (Table 1). Some differences in productivity among sites are undoubtedly related to differences in precipitation and climate (Table 1). However, differences in productivity are difficult to assign to climatic differences alone. We would, however, expect site-specific estimates of basal area to be related to productivity region, and they generally were (Table 1). The Laughry Bluff site was damaged by a wind storm 2 weeks prior to measurement and consequently had an uncharacteristically low basal area. Excluding the Laughry Bluff site, the correlation between basal area and the regional productivity estimates was 0.74.

### Statistical Methods

We used standard descriptive statistics to establish the baseline means and variances for this long-term study. Coefficient of variation (CV) was used to compute relative variability among the 10 sites for each measured variable.

Linear and nonlinear regression were used to quantify the relation between forest floor characteristics and the regional productivity gradient and also between leaf litter cover percent and litter weight. Spearman rank correlation was also used to test associations between productivity class and forest floor cover.

## RESULTS

### Percent Cover

Mean percent cover was greatest for dead leaves and herbaceous vegetation at 85% and least for exposed rock at 1% (Table 2). Across the 10 sites percent cover of bare ground had the greatest variation with a CV of 164%. Percent cover of dead leaves and dead herbaceous vegetation showed the least variation across the region with a CV of 12%.

Table 2.—Percent cover and litter weight means, standard deviations (SD) and coefficient of variation (CV) percent for the entire three-state region

| % cover of: | Mean | SD | CV (%) |
|---|---|---|---|
| Dead leaves and dead herbaceous vegetation | 85 | 10 | 12 |
| Bare Ground | 6 | 10 | 164 |
| Downed woody material | 5 | 1 | 26 |
| Boles of woody vegetation >= 1 m tall | 2 | 1 | 63 |
| Exposed rock | 1 | 2 | 157 |
| *Other | 1 | 2 | 132 |
| **% cover of aerial component:** | | | |
| Live herbaceous cover | 32 | 22 | 69 |
| **Litter weight:** | | | |
| Litter (tons/ac) | 7 | 3 | 46 |
| Weight ratio (NE/SW slopes) | 0.99 | 0.16 | 16 |

*Other = mostly moss

Litter Weight

The mean for litter weight for all 10 sites was 6.8 tons/ac (15.4 metric tons/ha) and was highly variable among sites with a CV of 46%. The mean ratio of litter weight (NE/SW slopes) was 0.99 and had the second smallest CV at 16% (Table 2).

Variability of Percent Cover and Litter Weight Across the Productivity Gradient

For most percent cover variables, the relation to the productivity gradient was weak (Table 3 and Figure 1). However, the inverse relationship of percent cover of exposed rock to regional productivity class was highly significant ($R^2 = 0.88$ for negative exponential model, p-value < 0.01) (Table 3 and Figure 1a). Given the approximate nature of the relationship between productivity classes and individual old-growth sites, we also tested the association between exposed rock and the productivity using only rank order of each variable. The Spearman rank correlation was -0.91 ($p < 0.01$). The positive linear relation between litter weight and productivity class was also significant ($p = 0.01$, $R^2 = 0.55$) as was the rank correlation (Rho = 0.71, $p = 0.02$) (Figure 1b).

Although, trends in percent cover with productivity class were weak for most variables, some interesting patterns stand out. For example Engelman Woods (EW) and Spitler Woods (SW) show percent cover values very different than the other sites (Figure 1c and Figure 1d: percent cover of dead leaves and dead vegetation, percent cover of bare ground). In the litter weight graph (Figure 1b), SW has a litter weight that is less than expected based on its productivity region. Both EW and SW have overstories of sugar maple (*Acer saccharum)* while seven of the eight other sites have overstories dominated by oak-hickory (*Quercus-Carya*). In a study by McClaughrety and others (1985) the percent of initial dry matter of litter was greater for the white oak litter type than the sugar maple litter type after both 1 and 2 year intervals, when each litter type was observed in their respective stand types. Of the eight non-maple sites only Laughery Bluff (LA) does not have an overstory dominated by oak-hickory. LA has an overstory dominated by American beech (*Fagus grandifolia*) and yellow poplar (*Liriodendron tulipifera).*

Table 3.—$R^2$ and p-values for simple linear regression of each listed dependent variable where the independent variable is regional productivity class.

| Dependent Variable, % cover of: | $R^2$ | p-value |
|---|---|---|
| Dead leaves and dead herbaceous vegetation | 0.12 | 0.31 |
| Bare Ground | 0.03 | 0.63 |
| Downed woody material | 0.24 | 0.15 |
| Boles of woody vegetation >= 1 m tall | 0.15 | 0.28 |
| Exposed rock | 0.76 | 0.00 |
| **Dependent Variable, % cover of aerial component:** | | |
| Live herbaceous cover | 0.11 | 0.35 |
| **Dependent Variable, weight:** | | |
| Litter (tons/ac) | 0.55 | 0.01 |
| Weight ratio (NE/SW slopes) | 0.00 | 0.90 |

In an effort to further explain site conditions we explored potential relationships with basal area (BA). We examined total BA and BA of sugar maple at each site as well as all variables listed in table 3. Neither total BA or BA of sugar maple were as strongly related to site variables as was regional productivity.

Predicting Litter Weight

In these forests, dead leaves were the largest component of forest litter, and we expected to find a significant relationship between percent cover of dead leaves/dead herbaceous vegetation and litter weight. A simple linear model explained 69% of the variation in litter weight among sites as a function of percent cover of dead leaves and dead herbaceous vegetation.

Litter Weight (dry tons/ac) = -14.2657 + 0.2495*(percent cover) [1]

$R^2 = 0.69$ p-value < 0.01 root MSE = 1.85 n = 10

Where percent cover is cover of dead leaves and dead herbaceous vegetation measured as a percent (0-100)

We also know from Table 3 that part of the variation in litter weight can be explained by the regional productivity gradient. Model 2 illustrates the advantage of including both percent cover and regional productivity in the model. A linear model with terms for both percent cover and productivity region explained 92% of the variation in litter dry weight.

Litter Weight (dry tons/ac) = -17.239 + 0.1033*(productivity) + 0.1996*(percent cover) [2]

$R^2 = 0.92$ p-value < 0.01 root MSE = 1.03 n=10

Where: Productivity is in ($ft^3$/ac/yr) units as indicated in Table 1, and percent cover is as described in [1]

Both dependent variables are highly significant ($p < 0.01$).

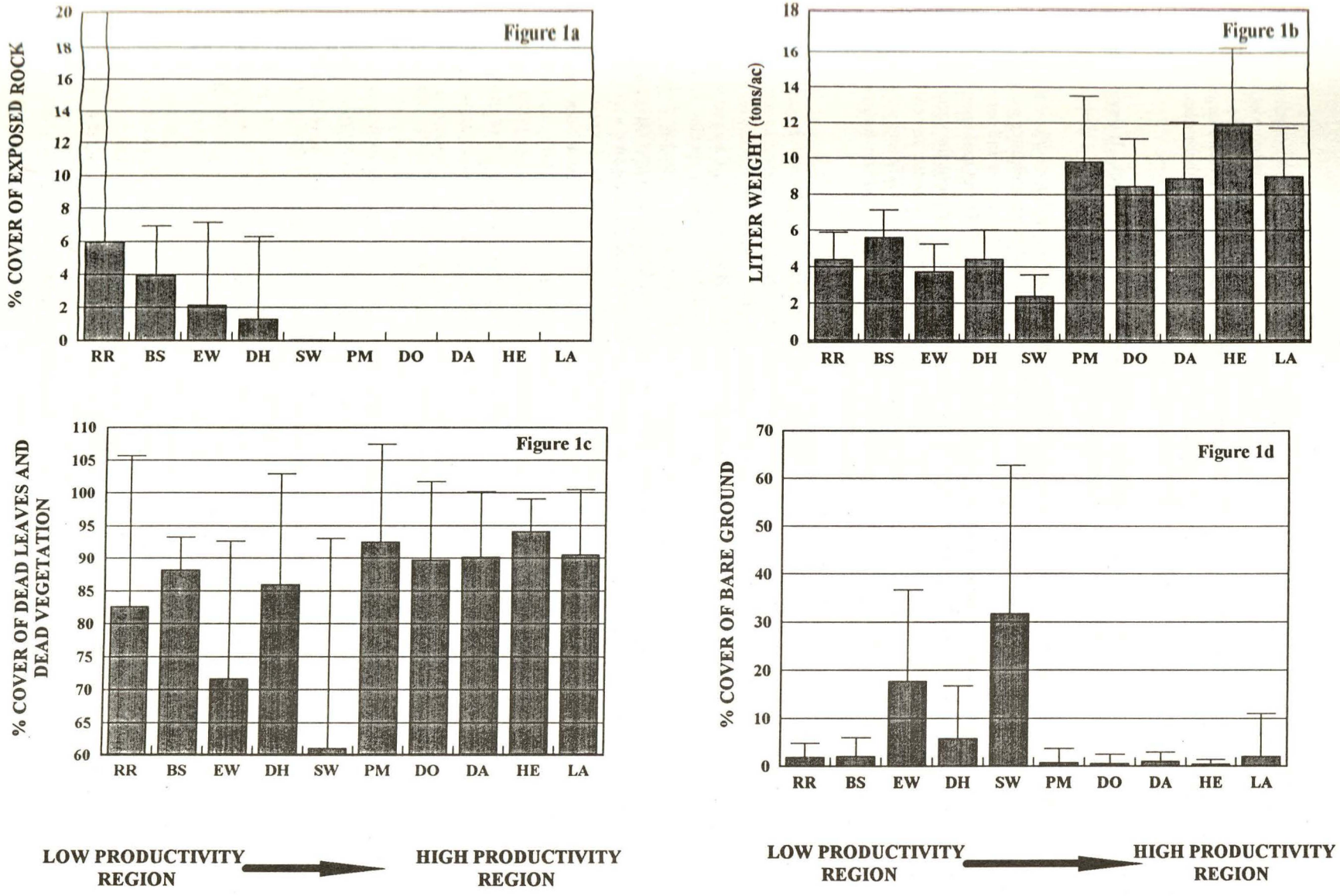


Figure 1.—Mean values and error (1 SD) for percent cover by sample site. Sites are ordered from lowest to highest regional productivity class. Site codes are given in Table 1.

## DISCUSSION

It is not surprising that the majority of the forest floor was covered by dead leaves and herbaceous matter. However, the fact that the percent coverage of this variable was 85% and more than 14 times greater than the next largest component is notable. Moreover, the coefficient of variation for percent cover of leaves and dead herbaceous material was lower than for any other variables we measured.

The variable with the strongest relationship to regional productivity class was percent exposed rock. It is an indicator of the shallow, droughty soils at the southern Missouri sites.

We found a strong linear relationship between percent leaves and herbaceous cover and litter dry weight. Although our model for litter weight estimation describes the form of the relationship between leaf cover and litter weight, the data included only sites with leaf cover exceeding 60%. Additional data over a wider range of sites and percent cover values could change the shape of the relationship as it would pass through the origin. Direct measurement of litter weight is still the most reliable inventory technique until we develop a better understanding of the variability of litter weight as these forests change through time. However, the strength of the relationship between percent leaf cover and litter dry weight suggests that some form of ratio or regression estimator would be an efficient way to measure litter weight at a specific site. Such an estimator would utilize a large number of inexpensive estimates of percent cover and a small number of relatively expensive litter weight estimates.

Many of these old-growth sites are expected to undergo changes in overstory species composition which could dramatically change the forest litter dynamics. Many old-growth woodlands have overstories of *Quercus* species with understories of tolerant species such as *Acer saccharum* and *Fagus grandifolia* (Lindsey 1962, Johnson and others 1973, Schlessinger 1976, McGee 1984, McGee 1986, Ward and Parker 1989, Boerner 1991, Cho and Boener 1991, Shotola and others 1992). Others have reported regeneration dominated by these understory species (Cain 1935, Quarterman and others 1972, Abrams and Downs 1990, Boerner 1991). The use of fire, grazing, and clearing for agriculture has a long history in the region which promoted domination of forests by seral species (Parker 1993, Olson 1996). Native Americans were active in the use of fire for agricultural clearing and driving game which is the likely cause of the seral species in canopies of many old-growth forests in the region (DenUyl 1954, Campbell 1989, DeVivo 1990, Reich and others 1990, Denevan 1992). Presettlement old-growth oak forests may have been maintained by fire, however modern fire suppression has been used in the Midwest for nearly a century (Rusterholz 1991). In a review of historical data dating back to the early part of this century McCune and Menges (1986) found that old-growth forests in the region had a strong successional trend toward shade tolerant species such as sugar maple and American beech.

In addition to affecting the species composition of the overstory, changes in wildfire regimes also directly affect the quantity of lief litter. The period 1820 to 1940 which was characterized by a fire-free interval of only 4 years for the vicinity of the Big Spring site (Guyette 1995) certainly resulted in lower litter weights at Big Spring than exist today. The physical presence or absence of leaf litter alters the environment of germinating plants. In addition to altering seed bed characteristics, in some cases abundant leaf litter can act as a physical barrier to plant growth (Sydes and Grime 1981). Reductions in fire frequency may increase the sheer physical impact of litter mass relative to the period when the overstory trees in these old-growth forests were established.

We found the litter weight of northeast facing vs. southwest facing slopes within these old-growth forests to have nearly a 1- to 1-ratio across the three-state area. This surprised us because of vegetation differences between northeast and southwest facing slopes. We would expect northeast slopes to be more productive with a higher rate of litter input than the southwest facing slopes. However litter in the cooler, moister northeast slopes could also be decomposing more rapidly. If this is the actual relationship, then an equilibrium of litter inputs and decomposition is a process that may be occurring regardless of aspect in these old-growth sites. This hypothesis could easily be tested using litter traps and by measuring mineralization rates over several growing seasons. Finally, overstory composition and site conditions should be considered when interpreting and applying the results of our inventory and analysis. The majority of our study sites were upland oak forests. Some of the largest anomalies in the general

trends we observed were associated with the two sites with overstories dominated by sugar maples. For instance, the rapid decomposition of sugar maple leaves and moist conditions at the Spitler Woods site were likely factors in the different than expected values there.

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# DISTURBANCE-RELATED CHANGES IN GROUND FLORA OF WEST VIRGINIA OAK FORESTS

R.M. Muzika, D.L. Feicht and S.L. Fosbroke[1]

While the effects of disturbance such as insect defoliation have been well documented in terms of overstory mortality and woody species regeneration, little is known about such disturbance effects on herbaceous flora, or non-commercial woody species. Since ground vegetation is often more sensitive to site and environmental conditions, such information may be valuable in understanding successional development in these forests and can be used as an indication of forest resilience and forest change. Furthermore, there exists some controversy about the effect of thinning on understory vegetation. We have studied vegetation in the lower structural layer for several years on oak/mixed hardwood stands that had been defoliated by gypsy moth (*Lymantria dispar* L.) in 1990 and 1991. Some of these stands were thinned in 1990 and also there were also control stands with neither thinning nor defoliation. We sampled herbaceous vegetation in 1992, 1993, and 1995. Estimates of other ground cover species such as *Rubus*, ferns, and grasses were determined in 1989, 1990, 1991, 1992, 1994, and 1996. Increases in species richness were evident in thinned stands. Using detrended correspondence to examine herbaceous vegetation patterns, we found that defoliation has a greater influence than thinning in separation of stands. However, the separation was greatest on DCA axis 2, suggesting that factors other than disturbance more strongly control the herbaceous vegetation. Site factors and original overstory composition tend to determine the composition of the ground vegetation. Coverage of competing vegetation (e.g., grasses and ferns) was greater than in stands that were only defoliated, suggesting that some caution is advisable when using silviculture as a management tool for gypsy moth.

[1]Research Ecologist, Forester, and Forester, respectively, Northeastern Forest Experiment Station, USDA Forest Service, 180 Canfield St., Morgantown, WV 26505.

# CHANGES IN DOWN DEAD WOOD VOLUME ACROSS A CHRONOSEQUENCE OF SILVICULTURAL OPENINGS IN SOUTHERN INDIANA FORESTS

Michael A. Jenkins and George R. Parker[1]

Abstract: The volume and decay stages of down dead wood were evaluated across a chronosequence of 46 silvicultural openings and 10 uncut control stands to determine how down dead wood volume changes with stand development. Openings ranged in age from 8 to 26 years and were divided into three age groups: (1) < 12 years, (2) 12-16 years, and (3) > 16 years. Individual logs were assigned to 5 decay stages ranging from freshly fallen (stage 1) to near total decomposition (stage 5). The volumes of down dead wood in age groups 1 and 2 were greater than that of the control plots (80-100 years old) and decreased with increasing stand age. This trend held true for both large and small diameter material. Down dead wood from decay stages 1 and 5 was scarce in all opening age groups. Age group 1 and 2 plots contained significantly greater volumes of moderately decayed material (decay stages 3 and 4) than the control stands. These findings and those of other researchers suggest that after an initial post-cutting pulse of down dead wood, volume decreases steadily throughout the first 26 years after cutting and levels off at 80 years. Comparison with other research suggests that mid-successional stands, such as the control stands in this study, have greatly reduced down dead volume compared to old growth stands.

## INTRODUCTION

Traditional studies of the effects of forest management on stand structure have emphasized vegetative structure, species composition, and species regeneration. The recent emphasis on ecosystem management has focused interest on how forest management affects forest components not traditionally studied, such as down dead wood (DDW) volume. However, studies of the effects of forest management on down dead wood volume in the Central Hardwood Region are lacking.

Research has shown the DDW is important in providing wildlife habitat, creating plant regeneration niches, and releasing nutrients through time. Studies suggest that down dead wood plays an important role the life cycles of numerous forest animals. Among small mammals, the family Soricidae (shrews) use DDW not only as protective cover, but as nesting sites and feeding areas (Harmon and others 1986). Rodents, such as red squirrel (*Tamiasciurus hudsonicus*), eastern gray squirrel (*Sciurus carolinensis*), and eastern chipmunk (*Tamias striatus*) also use woody debris for protective cover by hiding and nesting in hollow logs (Barkalow and Soots 1965, Burt and Grossenheider 1976).

Barnum and others (1992) found that white footed mice (*Peromyscus leucopus*) in a Maryland hardwood forest used logs as a means of traversing their home range. The mice avoided traveling on logs < 5 cm diameter, and preferred logs that were rotten or covered with moss, since large decayed logs may allow mice and other small mammals to make less noise when traveling, thereby reducing their risk of predation. In addition, the loosened bark of decaying logs offers thermal protection and nesting sites for salamanders and other species of amphibians and reptiles (Oliver 1955, Maser and Trappe 1986).

Down dead wood may play an important role in providing safe sites for woody species regeneration, although few if any eastern hardwood plant species are down dead wood obligates (Harmon and others 1986). Palmer (1987) found that while fallen logs and tip-up mounds comprised only 2.1% of forest floor area in a North Carolina old-growth forest, they were sites for 22.3% of tree species regeneration. Buckner and McCracken (1978) found that tip-up

[1]Graduate Instructor and Professor, respectively, Department of Forestry and Natural Resources, 1159 Forestry Building, Purdue University, West Lafayette, IN 47907-1159.

mounds created by fallen trees provide important regeneration niches for tulip poplar (*Liriodendron tulipifera* L.) in climax forest stands. While decomposing logs generally have lower nutrient levels than surrounding soil (Harmon and others 1986), they can provide regeneration sites free from over-shading by shrubs (Palmer 1987), buffer against extreme temperatures (Harmon et al. 1986), and protect seedlings from material moving down slope (Harmon and others 1986).

Organic material in the form of snags, fallen trees, and decaying logs exert considerable ecological control on a site for decades, or even centuries (Van Lear 1993). This influence results from the relatively slow decay rate of large DDW in hardwood forests. McFee and Stone (1966) observed that yellow birch (*Betula allephaniensis*) DDW persists for a century or more in mature northern forests. In an Indiana old-growth forest, MacMillan (1988) observed that *Quercus* logs decayed at a rate of 1.8% per year. Using this rate of decay, a 3 meter log with a diameter of 30 cm would require 39 years to lose 50% of its mass and 130 years to lose 90% of its mass. As DDW decomposes, nitrogen and other nutrients are sequestered by microorganisms and later become available for plant growth (MacMillan 1981). While the concentration of nutrients stored in woody material is low relative to that in leaves and fine roots, DDW may still be an important component of the biological and biochemical nutrient cycles.

Past studies in northern (Gore and Patterson 1986) and southern (McCarthy and Bailey 1994) Appalachian hardwood forests have examined DDW biomass in clearcuts, mature stands, and old-growth forests. These studies found that DDW biomass is highest in recent clearcuts, declines in mature forest, and increases in old-growth forests. Changes in total DDW volume, volume by decay stage, and volume by diameter class as a response to forest management have not been examined in the Central Hardwood Region.

We will address three main questions in this paper. First, how does total DDW volume vary with stand succession stage? Second, how does the DDW volume of different decay stages vary with stand age? Finally, how does the diameter of down dead wood vary with successional stage? We will examine how these parameters vary in 8-26 year-old silvicultural openings and 80-100 year-old mature stands. We will also compare the results of our study to those of an old-growth study also conducted in southern Indiana.

## METHODS

Between 1994 and 1996 we sampled 46 silvicultural openings (commercial group selections and clearcuts) and 10 uncut control stands on the Pleasant Run Unit of Hoosier National Forest, Morgan-Monroe State Forest, and Yellowwood State Forest in southern Indiana. To remove site induced variation, all plots were located on *Fagus-Acer saccharum/Arisaema* mesic slopes as delineated by Van Kley and others (1994). We sampled three silvicultural opening age groups; (1) < 12 years, (2) 12-16 years; and (3) > 16 years. All totaled, we sampled 16 age group 1 plots, 19 age group 2 plots, 11 age group 3 plots, and 10 control plots (age group 4). Openings ranged from 8 to 26 years in age and from 0.1 to 11 ha in size. Due to public pressure, there has been no major cutting on the Hoosier National Forest since 1986. Therefore, our study did not include any openings less than 8 years in age. Control plots were all 80-100 years old.

### Field Sampling

We estimated DDW volume within a 500m$^2$ plot placed within the silvicultural opening and control stand plots. We placed the plot within the center of smaller openings and randomly along the mesic slope in larger openings and control stands. Within each plot, we measured the midpoint diameter and length of all down dead wood 10 cm or greater in diameter.

Each DDW segment was visually classified into one of five decay stages (Thomas 1979). Decay stages ranged from 1 (freshly fallen) to 5 (near complete decomposition). If a piece of down dead wood contained more than one decay stage, we measured it as separate segments by decay stage.

Data preparation and analysis: Volume was determined in $m^3$ with the following equation for the volume of a cylinder:

$$[(\text{Midpoint diamter}/2)^2\pi] * \text{Segment length}$$

The mean DDW volume of each group class was calculated. Within each age group, volume was further subdivided by decay class and diameter class for comparison. Four diameter classes were compared: (1) 10-20 cm, (2) 21-30 cm, (3) 31-40 cm, and (4) > 40. All values were converted to a per hectare basis and compared between age groups using one-way Analysis of Variance and the Tukey test for pair-wise multiple comparisons. Changes in DDW volume through time was analyzed with regression analysis.

## RESULTS

Our results show distinct differences in the total DDW volume of the silvicultural openings and control plots (Figure 1, Table 1). The two youngest age groups (1 and 2) contained greater volumes of down dead wood than the older groups 3 and 4. Plots located in openings (age groups 1-3) contained significantly greater volumes of DDW than the uncut control plots (group 4, Table 1). Age group 2 plots contained the greatest amount of stage 4 down dead wood, suggesting that at least 12-16 years were required for logs to decay to this stage. Since we lacked preharvest DDW data, the preharvest DDW volume of these plots is unknown. However, our age class 4 data suggest that the DDW volume in precut stands was probably relatively low. Therefore, most of the DDW observed in our plots was probably post-harvest in origin. The volume of decay stage 4 DDW was greatly reduced in age group 3 plots and further reduced in age group 4 (control) plots. Decay stage 3 comprised the vast majority of DDW volume across all age groups. This suggests that decay stage 3 may need to be subdivided into several smaller stages to better assess down dead wood dynamics. Decay stage 5 DDW was nearly absent from plots of all four age groups, and decay stage 1 was absent from all four age groups.

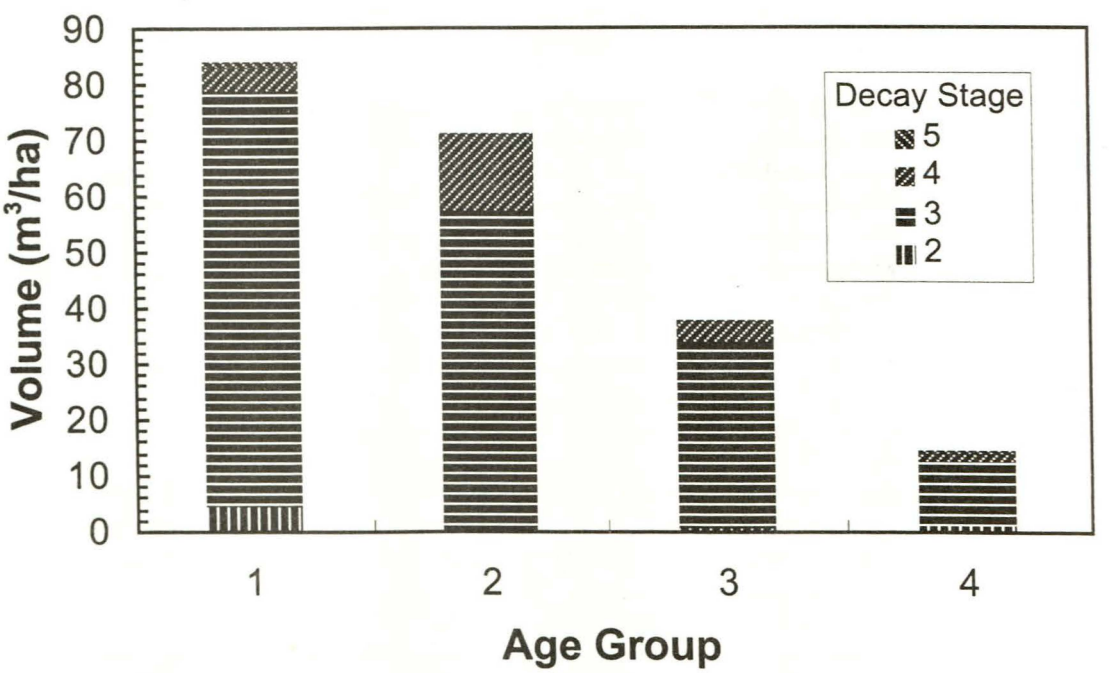


Figure 1. Down dead wood volume of all age groups by decay stage. Age group (1) < 12 years, (2) 12-16 years, (3) >16 years and (4) 80-100 years (control plots). Decay stage 1 is the least decayed and decay stage 5 is the most decayed. No decay stage 1 logs were found on the study plots.

Table 1. Comparison of mean volume ($m^3$/ha) by decay stage for all age groups. Age group (1) < 12 years, age group (2) 12-16 years, (3) >16 years, and (4) 80-100 year old control stands. Decay stage 1 is the least decayed and decay stage 5 is the most decayed. No decay stage 1 logs were found on the study plots. Means within a row not superscripted with the same letter are significantly different ($P < 0.05$).

| | Age Group 1 | | Age Group 2 | | Age Group 3 | | Age Group 4 | |
|---|---|---|---|---|---|---|---|---|
| | Mean | SD | Mean | SD | Mean | SD | Mean | SD |
| Decay Stage 2 | 4.08[a] | 12.47 | 0.04[a] | 0.11 | 0.50[a] | 1.21 | 0.75[a] | 1.59 |
| Decay Stage 3 | 74.27[a] | 39.69 | 57.25[ab] | 39.77 | 33.37[bc] | 24.40 | 11.60[c] | 11.80 |
| Decay Stage 4 | 4.62[b] | 4.54 | 13.44[a] | 8.99 | 3.93[b] | 3.51 | 2.06[b] | 2.70 |
| Decay Stage 5 | 0.67[a] | 1.88 | 0.05[a] | 0.14 | 0.00[a] | 0.00 | 0.27[a] | 0.86 |
| Total | 83.64[a] | 33.61 | 70.77[ab] | 42.40 | 37.80[b] | 24.71 | 14.71[c] | 11.29 |

Our results also suggest that the rate of change in down dead wood volume is not constant across age classes (Figure 2). Regression analysis with a negative exponential model suggested that volume decreased rapidly with increasing stand age, and leveled off in 80-100 year stands. Following harvest, smaller diameter slash decays quickly, resulting in a rapid decrease in DDW volume. Our study contained only 8-26 year old openings and 80-100 year stands. Additional sampling of 0-8 year old and 30-70 year old stands is needed to help clarify changes in DDW volume in these forests.

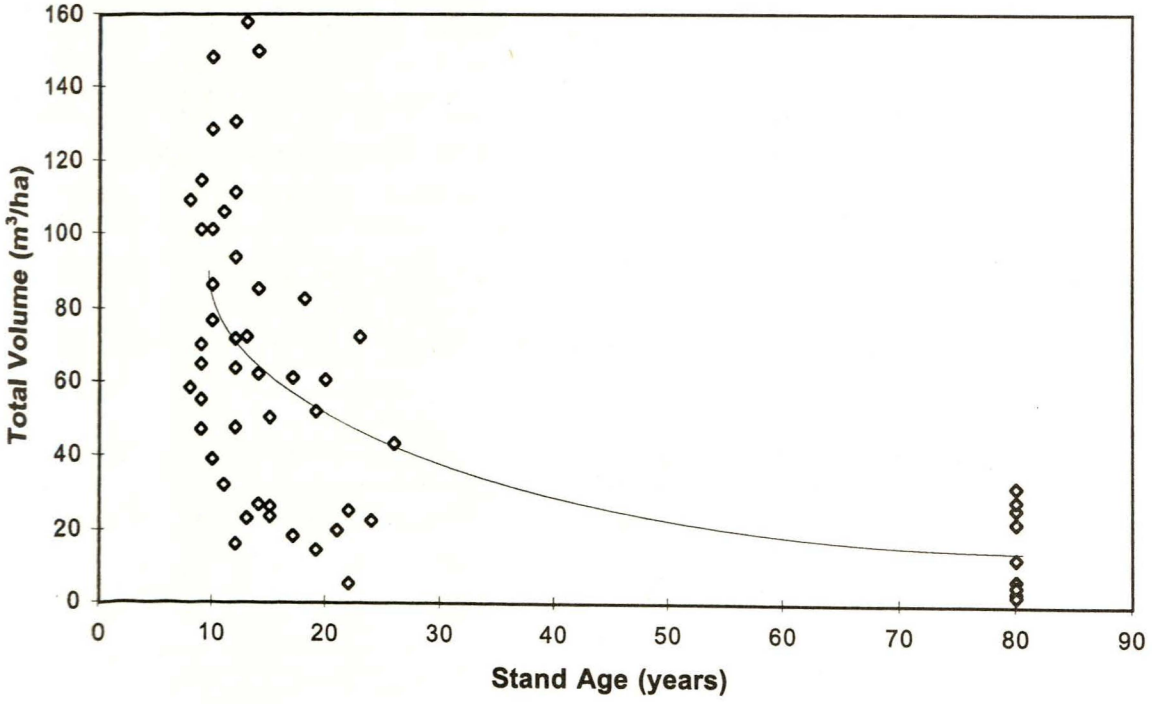


Figure 2. Negative exponential model of stand age vs. total down dead wood volume. Total volume = $146.62e^{0.06Age}$. $R^2 = 0.67$, Adjusted $R^2 = 0.45$, $P < 0.001$.

The volume of DDW across the four age groups varied by diameter class (Figure 3). The older age groups (3 and 4) had much lower volumes of DDW for all 4 diameter classes. Age group 1 plots contained significantly greater volumes of all four diameter classes of DDW than age group 4 plots (Table 2). Generally, the volume of DDW from all four diameter classes decreased with increasing stand age. The control plots (age group 4) were almost entirely lacking in larger diameter DDW, suggesting that most DDW on the sites was from smaller diameter fallen branches, instead of whole fallen trees.

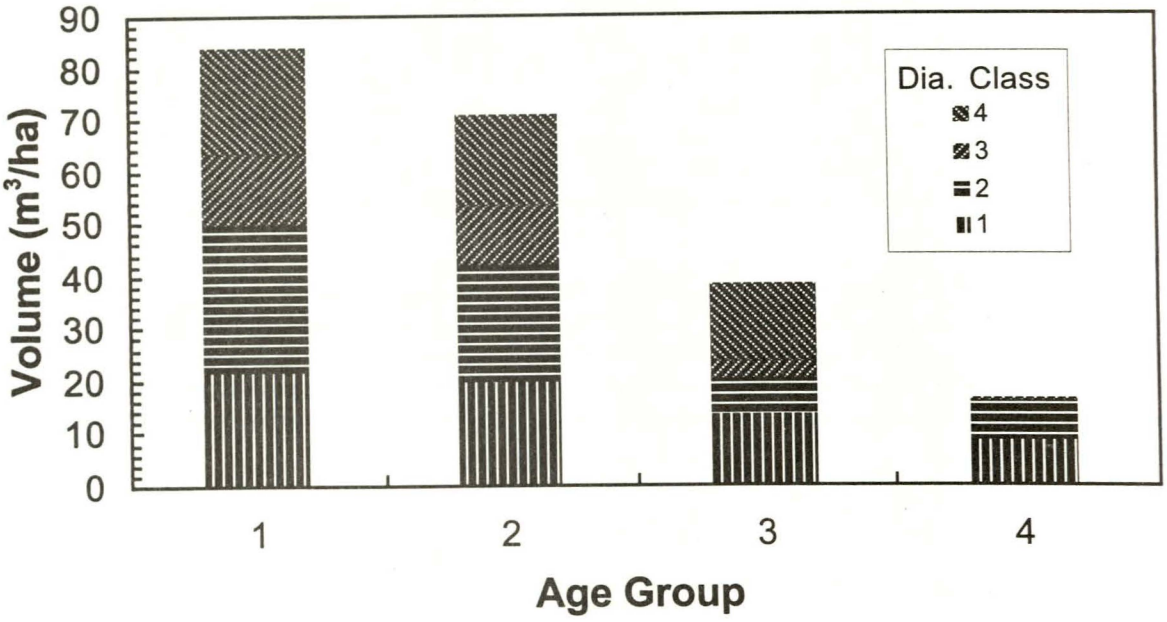


Figure 3. Down dead wood volume of all age groups by diameter class. Age group (1) < 12 years, age group (2) 12-16 years, and (3) >16 years. Diameter class (1) 10-20 cm, (2) 21-30 cm, (3) 31-40 cm, and (4) > 40cm.

Table 2. Comparison of mean volume ($m^3$/ha) by diameter class for all age groups. Age group (1) < 12 years, (2) 12-16 years, (3) >16 years, and (4) 80-100 year old control stands. Diameter class (1) 10-20 cm, (2) 21-30 cm, (3) 31-40 cm, and (4) > 40cm. Means within a row not superscripted with the same letter are significantly different ($P < 0.05$).

| | Age Group 1 | | Age Group 2 | | Age Group 3 | | Age Group 4 | |
|---|---|---|---|---|---|---|---|---|
| | Mean | SD | Mean | SD | Mean | SD | Mean | SD |
| Diameter Class 1 | 21.40[a] | 11.32 | 19.58[a] | 13.15 | 12.92[ab] | 10.93 | 7.05[b] | 4.44 |
| Diameter Class 2 | 28.02[a] | 20.25 | 22.30[ab] | 18.43 | 7.05[b] | 8.17 | 7.20[b] | 8.05 |
| Diameter Class 3 | 13.86[a] | 14.64 | 10.95[ab] | 14.51 | 3.54[ab] | 6.18 | 0.46[b] | 0.86 |
| Diameter Class 4 | 20.35[a] | 23.18 | 17.94[a] | 19.18 | 14.29[a] | 22.41 | 0.00[a] | 0.00 |

## DISCUSSION

The results of our study exhibit trends similar to studies conducted in other parts of the eastern hardwood forest. Gore and Patterson (1986) also observed sharp decreases in down dead wood following harvest in New Hampshire hardwood forests and McCarthy and Bailey (1994) observed similar results in southern Appalachian forests. Gore and Patterson (1986) observed much lower down dead wood volumes in 50 year stands. The high volume of DDW present in recent silvicultural openings results from the addition of post-harvest slash and stumps. While our study lacked data from openings 1-8 years in age, these younger openings tend to contain even greater volumes of less-decayed post-harvest slash (McCarthy and Bailey 1994). The majority of this input is of relatively small diameter and has a large surface area that leads to rapid decomposition. Once the majority of this small diameter material decays, a lag period occurs because the small volume of down dead wood input from the developing stand is less than the loss from decomposition (Figure 2). We observed little input of DDW from young freshly fallen trees in age classes 1-3. While mortality does occur in these developing stands, most dead trees are less than 10 cm diameter and therefore were not measured in this study. This volume of DDW is probably negligible compared to the volume of post-harvest slash.

The 80-100 year old stands (age group 4) are in the understory reinitiation stage as defined by Oliver (1981). At this stage of stand development, there is little overstory mortality, and therefore little input of DDW in the form of dead trees. Age class 4 plots (80-100 years old) had a low mean relative density of dead trees greater than 30 cm dbh (relative density = 3.5, standard error = 1.87). This low mortality explains the generally low overall down dead wood volume in these stands and the low volume of highly decayed and large diameter dead wood. We expect that canopy trees will begin to die as these stands develop over time (McGee 1984).

Comparison of our data to those collected from an Indiana old-growth forest revealed several important trends. Spetich (1995) found a total down dead wood volume of 74.2 $m^3$/ha in Donaldson's Woods, an old-growth forest remnant in southern Indiana. Age group 1 plots in our study contained a greater volume of down dead wood (83.6 $m^3$/ha) than Donaldson's Woods. However, Donaldson's Woods exhibited a greater volume of DDW than either age group 2 (70.8 $m^3$/ha) or age group 3 (37.8 $m^3$/ha) plots. In addition, the down dead wood volume of Donaldson's Woods was approximately three times greater than that of our age group 4 (control) plots (15.8 $m^3$/ha).

The openings and control stands sampled in this study generally exhibited lower volumes of logs by decay stage than Donaldson's Woods. Decay stage 5 down dead wood volume was much greater in Donaldson's Woods (6.5 $m^3$/ha) than in any of the ages classes sampled in this study (maximum = 0.7 $m^3$/ha for age class 1). Donaldson's Woods also exhibited a greater volume of decay stage 4 down dead material (11.4 $m^3$/ha) than age groups 1, 3, and 4 (4.6 $m^3$/ha, 3.9 $m^3$/ha, and 1.9 $m^3$/ha, respectively). Age group 2 plots contained a greater volume of decay stage 4 down dead wood (13.4 $m^3$/ha) than Donaldson's Woods. This greater value was probably because of the decaying of post-harvest slash. In our study, decay stage 4 material had all but disappeared in age group 3 openings. This decay stage 4 material did not appear in these openings as decay stage 5 material. A possible explanation for the lack of decay stage 5 material is that the large volume of post-harvest slash may have decayed so quickly that it was not detected as decay stage 5 material. Spetich (1995) also observed a much greater volume of diameter class 4 (> 40 cm diameter) down dead wood. Age group 1 openings contained 20.4 $m^3$/ha of diameter class 4 down dead wood, compared to 43.5 $m^3$/ha for Donaldson's Woods. Age group 4 (control) plots contained no logs over 40 cm in diameter.

The large volume of highly decayed large diameter DDW has been offered as a characteristic of old-growth forests (Parker 1989, Spetich 1995). While old-growth forests in southern Indiana consist of small scattered fragments, the existance of forests across the landscape that contain some old-growth characteristics may be important in managing biodiversity. In addition, highly decayed down wood of large diameter has been found to be very important habitat for numerous animal species (Oliver 1955, Harmon and others 1986, Maser and Trappe 1986, Barnum and others 1992). In light of these studies, it appears that the mature 80-100 year forest matrix of southern Indiana may not provide optimum habitat for these species. More research is needed in the more recently cut stands, which provide greater volumes of larger diameter DDW, to determine their suitability for these species. If continued research supports the importance of down dead wood in the Central Hardwood Forest Region, management techniques may need to be modified to better allow for this forest component.

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# CHANGES IN FINE ROOT DYNAMICS AND DISTRIBUTION ALONG A CHRONOSEQUENCE OF UPLAND OAK-HICKORY FORESTS

Travis W. Idol[1], Phillip E. Pope[1], and Felix Ponder, Jr.[2]

Abstract: Central Hardwood forests regenerate rapidly following a major disturbance like a clear-cut. The subsequent aboveground growth and development of the resulting stands have been well-documented for these forests. However, the belowground components, specifically the dynamics of fine roots, are not well understood. Fine roots are the main vectors through which plants uptake water and nutrients and represent a significant proportion of a plant's net primary productivity. Studying the dynamics of fine roots during stand/forest development is essential to understand forest ecosystems and for responsible forest management.

This study was installed at the Southern Indiana Purdue Agricultural Center (SIPAC) in Dubois County, Indiana to assess the fine root dynamics in stands of different age. The stands are historically upland oak-hickory forests and each was clear-cut at a different time in the past. The first stand was cut in 1991 (Site 1), the second in 1985 (Site 2), the third in 1966 (Site 3), and the fourth is a mature stand (80-120 years old) which serves as a control (Site 4). Stand density, species composition, and litterfall assessments of the aboveground vegetation have been conducted in companion studies. In this study, fine root dynamics: biomass, growth, mortality, and decomposition, and the soil environment: temperature, moisture, and N and P availability were evaluated in both the A and E or B horizons from July 1995 to June 1996. Time Period in Table 1 refers to the month, starting with July 1995, which the measurements represent.

Results indicate that fine root biomass is not significantly different among the stands, but that the live component of fine roots changes dramatically during the growing season. Fine root growth, mortality, and decomposition generally decrease from the younger to the older stands (Table 1). Patterns of fine root dynamics are also seasonally distinct among the stands. Peak fine root growth rates occur later during the fall in the younger stands than in the older ones. The distribution of fine roots becomes more concentrated in the A horizon with increasing stand age. Soil nitrate and phosphate availability are the soil variables most strongly related to fine root growth and mortality; whereas, fine root decomposition is also responsive to changes in soil temperature (Table 2). The quadratic nature of the relationships indicates that nutrient deficiencies may stimulate fine root growth in the regenerating stands. In general, following a clear-cut harvest, reestablishment of fine root biomass occurs more quickly than leaf area. Therefore, before canopy closure, availability of and competition for belowground resources may in large part determine the productivity and composition of regenerating stands in the Central Hardwoods Region.

[1] Graduate Research Assistant and Professor, respectively, Department of Forestry & Natural Resources, Purdue University, West Lafayette, IN.

[2] Supervisory Research Soil Scientist, USDA Forest Service, North Central Forest Experiment Station, 208 Foster Hall, Lincoln University, Jefferson City, MO.

Table 1. Fine root dynamics integrated over 30-cm depth

| Site | Fine root parameter | Time Period 1 | 3 | 5 | 8 | 11 | Annual average |
|---|---|---|---|---|---|---|---|
| | | | | | | | (Mg/ha) |
| 1 | Growth | 1.42b | 3.09a | 3.48a | 1.80b | 1.62b | 7.96 |
| | Mortality | 1.59c | 2.99b | 4.33a | 1.86c | 1.26c | 8.87 |
| | Decomposition | 1.15b | -0.18b | 7.07a | 1.11b | 1.64b | 7.15 |
| 2 | Growth | 1.44c | 3.12b | 3.87a | 2.18c | 1.93c | 8.87 |
| | Mortality | 2.82ab | 2.95ab | 4.40a | 2.53bc | 0.78c | 9.60 |
| | Decomposition | 2.44bc | 0.74bc | 5.09a | 4.20ab | 0.15c | 10.11 |
| 3 | Growth | 1.84ab | 4.83a | 1.87b | 0.73b | 1.21b | 6.72 |
| | Mortality | 4.10ab | 5.84a | 1.96b | 0.50b | 1.00b | 8.32 |
| | Decomposition | 3.24ab | 4.94a | 2.31ab | 1.19b | 0.38b | 7.92 |
| 4 | Growth | 1.52b | 3.01a | 1.54b | 0.94c | 0.97c | 5.36 |
| | Mortality | 2.72b | 4.79a | 1.63bc | 0.62cd | 0.17d | 6.28 |
| | Decomposition | 1.99a | 2.69a | 1.81a | 2.09ab | 0.12b | 6.46 |

Time Period values expressed in g/m$^2$-day. Values followed by the same letter within a row do not differ significantly.

Table 2. Regression analysis of fine root parameters-single depth

| Site | Parameter | Regression equation | $R^2$ | P-value |
|---|---|---|---|---|
| 1 | Growth | No significant relationships | | |
| | Mortality | No significant relationships | | |
| | Decomposition | $Y = 36.541341 - 3.494705*Temp + 0.084657*(Temp)^2$ | 0 .973 | <0.05 |
| 2 | Growth | $Y = 3.938654 - 0.649512*NO_3 + 0.049107*(NO_3)^2$ | .998 | <.01 |
| | Growth | $Y = 4.017974 - 1.329603*Phos + 0.170152*(Phos)^2$ | .674 | <.01 |
| | Mortality | $Y = 3.069869 + 0.127808*NO_3 - 0.044552*(NO3)^2$ | .741 | <.05 |
| | Decomposition | $Y = 31.536046 - 3.428857*Temp + 0.094458*(Temp)^2$ | .981 | <.05 |
| 3 | Growth | $Y = 4.968778 - 4.229065*Phos + 1.101870*(Phos)^2$ | .882 | <.01 |
| | Mortality | $Y = 6.054396 - 5.790557*Phos + 1.638396*(Phos)^2$ | .926 | <.05 |
| | Decomposition | $Y = 5.627110 - 5.475538*Phos + 1.514369*(Phos)^2$ | .990 | <.01 |
| 4 | Growth | No significant relationships | | |
| | Mortality | No significant relationships | | |
| | Decomposition | No significant relationships | | |

# HISTORY OF FIRE IN A SOUTHERN OHIO SECOND-GROWTH MIXED-OAK FOREST

Elaine Kennedy Sutherland[1]

Abstract: The role of fire in shaping the composition and structure of *Quercus* (oak)-dominated communities in the deciduous forests of eastern North America is becoming clearer but fire regimes are less well known. I analyzed the fire-scar patterns in 14 oak cross sections from a mixed-oak stand in Vinton County, southeastern Ohio, to determine the frequency and seasonality of fire. This oak stand was even-aged and originated in the mid-1850s, so the fire history is representative of the period after Euro-American settlement. The tree rings were crossdated, calendar year of the fire scars was assigned, and season of burning noted. The fire-scar dates are Weibull-distributed, and the median probability interval is 3.6 years. The 95% and 5% Weibull probability exceedance intervals were 0.4 and 12.1. Spatially extensive (major) fires occurred every 7.5 years, and the 95% and 5% exceedance intervals for these fires were 0.9 and 24.3. The fires probably were ignited by people. Fire scars indicated that most fires (69%) occurred during the dormant season or in the spring (25%) and were rare (6%) during the summer (May 1 to July 30). These patterns are not significantly different from dates of observed wildfires in Vinton County. Seasonal climate has no apparent relationship to fire occurrence.

## INTRODUCTION

Fire has played an undeniable role in the development of past and existing oak (*Quercus*) forests in eastern North America (Abrams 1992, Olson 1996). Lorimer (1985) characterized fire-related features of oak: oaks resist damage and mortality from fire because they have thick bark and rapidly compartmentalize cambium damaged by fire. Fire creates seedbeds for acorn germination, and seedlings and young trees sprout readily after being topkilled. Thus, in a regime of periodic fire, oaks are favored over shade-tolerant trees, especially *Acer* (maple) species, that are susceptible to fire (Abrams 1992). Since about 1930, effective fire protection has been applied broadly to the eastern deciduous forest (Pyne 1984). Abrams (1992) summarized widespread evidence that in the absence of fire, oaks are likely to be replaced by shade-tolerant species.

While recurring fire clearly was associated with the development of present-day oak forests, the fire regimes of oak forests are ill defined. Gill (1975) described the elements of fire regimes as frequency, intensity, spatial extent, seasonality, and type of fire (e.g., ground, surface, or crown fire). In coniferous forests of the western United States, fire regimes have been characterized by examining fire scars in the annual rings of centuries-old trees, and developing chronologies of fire (e.g., Baisan and Swetnam 1990, Brown and Seig 1996, Dieterich and Swetnam 1984 ). There are few fire histories of eastern oak forests (e.g., Abrams 1985, Cutter and Guyette 1994, Guyette and Cutter 1991, Guyette and McGinnes 1982) primarily because little of the presettlement forest remains after extensive deforestation following Euro-American settlement (Williams 1989). However, fire was an important process in the development of existing second-growth oak forests (Abrams 1992), and any information gained about the fire regimes that resulted in present forest composition and structure would be valuable in managing oak forests.

The history of Ohio's oak-dominated forests is typical of that of many Midwestern forests. Ninety-five percent forested at the time of Euro-American settlement (circa 1800), Ohio's forested area was reduced to about 15% of land cover by 1900, but has regrown to 30% since 1940. Currently, 65% of the unglaciated Allegheny plateau in southeastern Ohio is forested, and oak species dominate the overstory (Griffith and others 1993). The objectives of this study were to develop a fire history from a second-growth oak site in southeastern Ohio, and to make inferences about the past fire regime that resulted in that oak forest.

[1]Research Ecologist, USDA Forest Service, Northeastern Forest Experiment Station, 359 Main Road, Delaware, OH 43015-8640.

## PHYSIOGRAPHIC AND HISTORICAL SETTING

The area studied was within the unglaciated Allegheny Plateau physiographic region of southeastern Ohio, referred to as the Hill Country (Braun 1950). The topography was complex and dominated by highly dissected ridge and valley complexes with relative relief of 90 to 150 m. The climate was cool, temperate, and continental. October was the driest month; otherwise, rainfall was distributed relatively evenly through the year (King 1979).

Of the forest types in the Hill Country, 70% are mixed oak (*Quercus alba* L. (white oak), *Q. velutina* Lam. (black oak), and *Q. rubra* L. (northern red oak) (Type 52, Eyre 1980)). These upland forests lie in a geographical transition between the open woodlands of the Midwest and the more mesophytic forests of the Appalachians (Braun 1950).

Settlement of Vinton County in Ohio's Hill Country by Euro-Americans began in 1802, and an agricultural economy developed, primarily in bottomlands, where people cleared for field crops and pastures by tree-girdling and burning (Beatley 1959). Many more people migrated into the area in the 1850's with the development of the charcoal pig iron industry (Willard 1916). The demand for this high-quality iron caused rapid deforestation of the area around the smelters. Vinton County was 75% forested in 1853 when the furnaces were built, 59% forested by 1870, and only 18% forested by 1883 (Beatley 1959). Pig iron production peaked in the 1880's, and then declined with the loss of the timber resource for charcoal. Most of the smelters were closed by the turn of the century. The communities that surrounded the smelters were abandoned and the forests regrew (Stout 1933).

The site described in this study, Oreton, (latitude 39°09' longitude 82°24') was a 8-ha compartment on the Raccoon Ecological Management Area owned by Mead Paper Corporation in southern Vinton County, Ohio. The site was north-facing, with slopes of 0 to 25%. The site was purchased by Mead in 1965 from Sentry Royalty Company, a mining concern. Timber maps from that time show this was an uncut stand of second-growth upland oaks and other hardwood tree species.[2] The stand was second-growth, and the overstory was dominated by even-aged oak species about 150 years old, primarily black oak, white oak, and chestnut oak (*Q. prinus* L.).

## METHODS

Analysis of fire scars initiated by low-intensity fire events is the most appropriate way to reconstruct fire history in surface fire regimes (Swetnam and Baisan 1996). The approach here was to collect samples opportunistically during harvesting activities with cooperating timber operators. Samples were collected for the fire-scar analysis during a selection cut conducted by Mead in September 1995. By prior arrangement, trees were cut 10 to 15 cm above the ground and the logs examined to determine whether cross sections would be cut from them. Trees were selected for fire-scar sampling if the logs were relatively free from center decay but showed some evidence of scarring; this is usually visible even on a rough, chain-sawed surface. Sampled trees were scattered throughout the 8-ha compartment. Twenty-eight intact crossections were identified and transported to the laboratory where they were allowed to dry for a month. After drying, the sections were examined, and 14 that showed promise for analysis (intact, and with multiple fire scars) were surfaced using a power hand planer and belt sander. (The remaining 14 sections had large rot pockets or other physical defects that made them unsuitable for further analysis.) Each section was crossdated (Fritts 1976, Swetnam and others 1985) using two lines drawn from the pith to the bark as close to 180° apart as possible, but at least 90° to 180° apart. The lines were skeleton plotted (Stokes and Smiley 1968) and these plots compared to master chronologies from the same area (WNF chronology in LeBlanc and Foster 1992, Sutherland, unpublished). Calendar-year dates were then assigned to each annual ring.

After the crossdating of all sections was verified, calendar-year dates were assigned to each fire scar. The position of each scar within or between annual rings (Baisan and Swetnam 1990, Dieterich and Swetnam 1984) was recorded when possible to estimate season of fires—dormant, earlywood (spring), or latewood (summer). The two dominant wildfire seasons on the Wayne National Forest in Ohio both occur during the dormant period (Haines and

[2]Personal communication 6/20/96, Steve Mathey, Mead Paper Corp., Fine Paper Division, Chillecothe, Ohio.

others 1975): March-April and October-November (61 and 32% of all fires, respectively). Because the majority of wildfires occur in the spring, for simplicity, all dormant-season scars were assigned to the later calendar year. For example, a dormant-season scar between the tree rings that grew in 1870 and 1871 was assigned to the year 1871.

The distribution of dormant-season, spring, and summer fire scars were compared to dates of wildfires from state and federal agencies in the Vinton County area using a $X^2$ analysis (Sokal and Rohlf 1981). From dendrograph data taken in 1958 in central Illinois, Fritts (1960) found that virtually all of the earlywood of white oak trees and two-thirds of the earlywood of northern red oak trees formed before leafout (April 11 to May 5). For simplicity, I designated the earlywood (spring) growth period to have been April 15 to 30, the latewood (late spring/summer) period to have been May 1 to July 31, and the dormant season to have been August 1 to April 14.

Fire frequency or fire-return interval is a measure of central tendency derived from fire history data that indicates how often fire tends to occur at a site. However, central tendency may be poorly estimated by statistics such as the (arithmetic) mean fire interval (MFI) because fire-interval data are rarely distributed symmetrically; that is, there is no upper bound for the maximum fire interval but the shortest possible fire interval is 1 year (Swetnam and Baisan in press). Because fire history data often are positively skewed (Baker 1992), more flexible distributions such as the Weibull provide a better fit to the data and more robust measures of central tendency than the MFI (Baker 1992, Clark 1989, Johnson 1992). The Weibull distribution can be used to estimate both central tendency and range of fire history data, expressed as exceedance probabilities (Grissino-Mayer 1995). The central tendencies of Weibull distributions are expressed as the 0.5 exceedance probability (the 50$^{th}$ percentile of the estimated distribution). This is called the Weibull Median Probability Interval WMPI. Range is expressed as the 0.95 and 0.05 exceedance probabilities, which can be interpreted as significantly short and long fire intervals (Grissino-Mayer 1995). The FHX2 program described by Grissino-Mayer (1995) compares the goodness-of-fit between the fire interval data and the normal and Weibull distributions with the one-sample Kolmogorov-Smirnov (K-S) test, and calculates the MFI, WMPI, and short and long exceedance probability. These values were calculated for all fires occurring at the site, and for major fires. Major fires were defined as years during which 25% or more of all samples or at least two samples were scarred. Fires probably were more spatially extensive and intense during major fire years than in years when only one tree was scarred (Guyette and Cutter 1991, Guyette and McGinnes 1982).

To compare the relationship between fire and climate, mean March and April precipitation was plotted for the period 1895 to 1995, and fire years were labeled according to the estimated season of the fire and the extent (major or minor) of the fire.

## RESULTS

The sampled stand, essentially even-aged, originated around 1856; no trees have a pith date later than 1864, so the fire history is representative of the post-settlement period only. Twelve of the fourteen trees analyzed are black oaks; one white oak and one chestnut oak complete the sample.

Here, fire scars are defined as wounds occurring low on the tree bole, identified by the presence of callus tissue and a growth response to the death of a section of cambium (Guyette and Cutter 1991). Occasionally, false rings were observed near the scar, apparently a response to wounding. Although Abrams (1985) used the presence of charcoal in the scar as a criteria for attributing fire as the cause for scarring, I am not. Presence of charcoal is a highly conservative criteria for denoting fire scars because heating to 60°C kills cambium (Kayll 1968) and causes wounding. Actual combustion of the cambium is not necessary to cause wounding, and in the low-intensity fires typical in oak woodlands, combustion is probably rare. Occasionally, scars occurred that did not show callus and growth response that I attribute to fire-wounding response; such scars were not included in the fire-scar record. Although scars can be caused by a number of sources, including small mammals, fungus, and insect attacks (Arno and Sneck 1977, Mitchell and others 1983), there was no evidence of physical abrasion or insect galleries associated with the (fire) scarring events. Nor are the scars likely the result of fungus attack. Examination of a cross section from this

site revealed that the broad "single year" nature of the wounds are not consistent with attack by a perthotrophic fungus as those in the genus *Armillaria* [3].

Many trees had more than one scar; the number of fire events causing scars ranged from one to seven per tree. Unlike conifers that typically develop a single scarred area or face with a number of scars and healing ridges (Baisan and Swetnam 1990, Dieterich and Swetnam 1984), scars were seldom visible from the outside. The area of damaged cambium generally was small (< 2 cm in length) and was compartmentalized rapidly and embedded in the stem within 3 to 5 years after the scarring event. Scars on oaks sampled at the Oreton site were not segregated in one area on the cross section, nor were scars usually near each other.

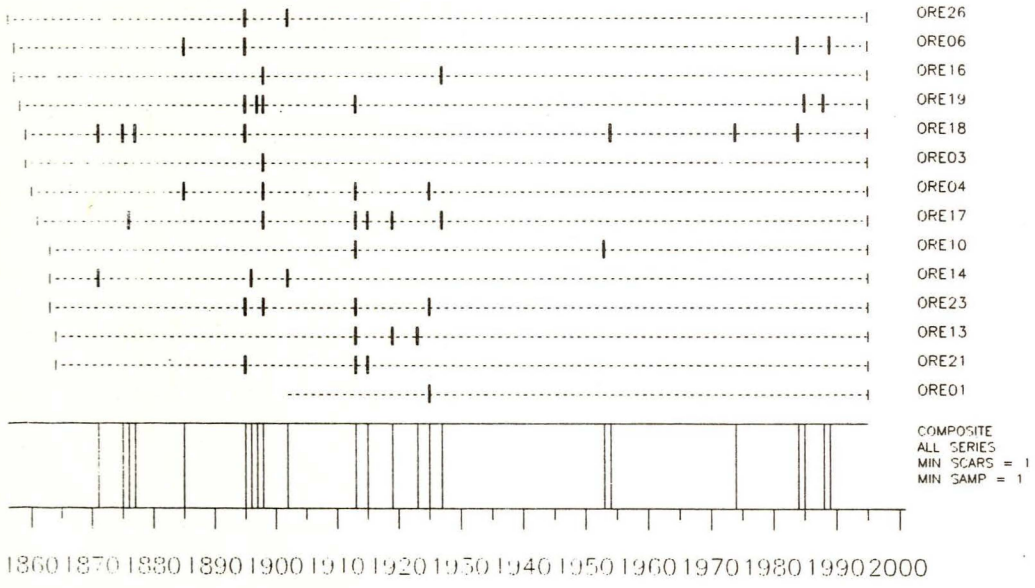


Figure 1. Fire chronology at the Oreton site, Vinton County, Ohio for the period 1856 to 1995. Each horizontal line represents the fire-scar chronology for an individual tree; each vertical bar represents a scar during a year.

Twenty-two fire events were recorded at the Oreton site (Figure 1, Table 1). The K-S d-statistic (0.573) was significant (Prob > d = 0.0001), so the data could not be modeled with a normal distribution. Alternatively, the Weibull distribution did model the fire-interval data adequately (K-S d-statistic = 0.222, Prob. > d = 0.2270). The observed fire intervals and calculated Weibull function for all fires and only major fires are shown in Figure 2. The median and the short and long probable interval values (Table 2) indicate that fire was likely to return to the Oreton site every 3 to 4 years, and that annual fires were unusual, as was an interval of more than 12 years between fires. Major fires (Table 2) occurred about half the time (every 7 to 8 years). It was unusual to have major fires in successive years, or for more than 24 years to pass between major fires.

[3]Personal communication, 11/12/96, Kevin Smith, Research Plant Physiologist, Northeastern Forest Experiment Station, Louis C. Wyman Forest Sciences Laboratory, PO Box 640, Durham, NH 03824-0640.

Table 1. Fire years, number of scars, fire interval, and season of fire of individual fires at the Oreton site, Vinton County, Ohio, 1856 to 1995. Fires occurred in 22 separate years. Major fires were designated where 25% of all samples or at least two cross sections had a fire scar during a given year. Seasons were designated as D = dormant, E = early spring (probably mid to late April), S = late spring/summer (probably May to mid July). (The * denotes significantly long ($p < 0.05$) intervals.)

| Year | Number of trees scarred | Fire interval (years) | Major fire interval (years) | Season |
|---|---|---|---|---|
| 1871 | 2 | . | . | D |
| 1875 | 1 | 4 | . | D |
| 1876 | 1 | 1 | . | D |
| 1877 | 1 | 1 | . | D |
| 1885 | 2 | 8 | 14 | D |
| 1895 | 6 | 10 | 10 | E |
| 1896 | 1 | 1 | . | D |
| 1897 | 1 | 1 | . | D |
| 1898 | 6 | 1 | 3 | E |
| 1902 | 2 | 4 | 4 | S |
| 1913 | 7 | 11 | 11 | E |
| 1915 | 2 | 2 | 2 | E |
| 1919 | 2 | 4 | 4 | D |
| 1923 | 1 | 4 | . | D |
| 1925 | 3 | 2 | 6 | D |
| 1927 | 2 | 2 | 2 | E |
| 1953 | 1 | 26* | . | E |
| 1954 | 1 | 1 | . | D |
| 1974 | 1 | 20* | . | S |
| 1984 | 2 | 10 | 57* | D |
| 1985 | 1 | 1 | . | S |
| 1988 | 1 | 3 | . | D |
| 1989 | 1 | 1 | . | D |

Most fires burned during the dormant season (69%; 33 scars) and during spring when the earlywood was forming (25.0%; 12 scars); only 6% (3 scars) burned during the summer when latewood was forming. The $X^2$ distribution of the scars during the dormant, spring, and late spring/summer seasons was not different ($X^2 = 1.82$, 2 d.f.) from wildfire occurrences in the Vinton County area (Figure 3). Thus, my estimates of the periods during which scars formed probably are accurate.

Figure 4 illustrates the relationship between fire and climate from 1895 to 1995. If climate conditions affect fire occurrence, then more fire events should occur on the bottom half of the graph (drier than normal conditions). There are nine fire events above the mean and nine below. Dormant-season fires usually occur in spring but can occur in autumn. Thus in Figure 4, autumn dormant-season fires would not be expected to show a relationship to spring climate and would only add error to climate/fire relationships in this graph. However earlywood fires do occur in the spring. Figure 4 shows that five earlywood fires happened when precipitation was average or above average but that only three occurred below the average level. Thus, there was no obvious relationship between precipitation and the occurrence of spring fire. This observation is consistent with that of Haines and others (1975), who counterintuitively found more spring fires in nondrought than in drought years on the Wayne National Forest. Conversely, they found more autumn fires during drought years than in nondrought years. The data available here cannot illustrate such relationships.

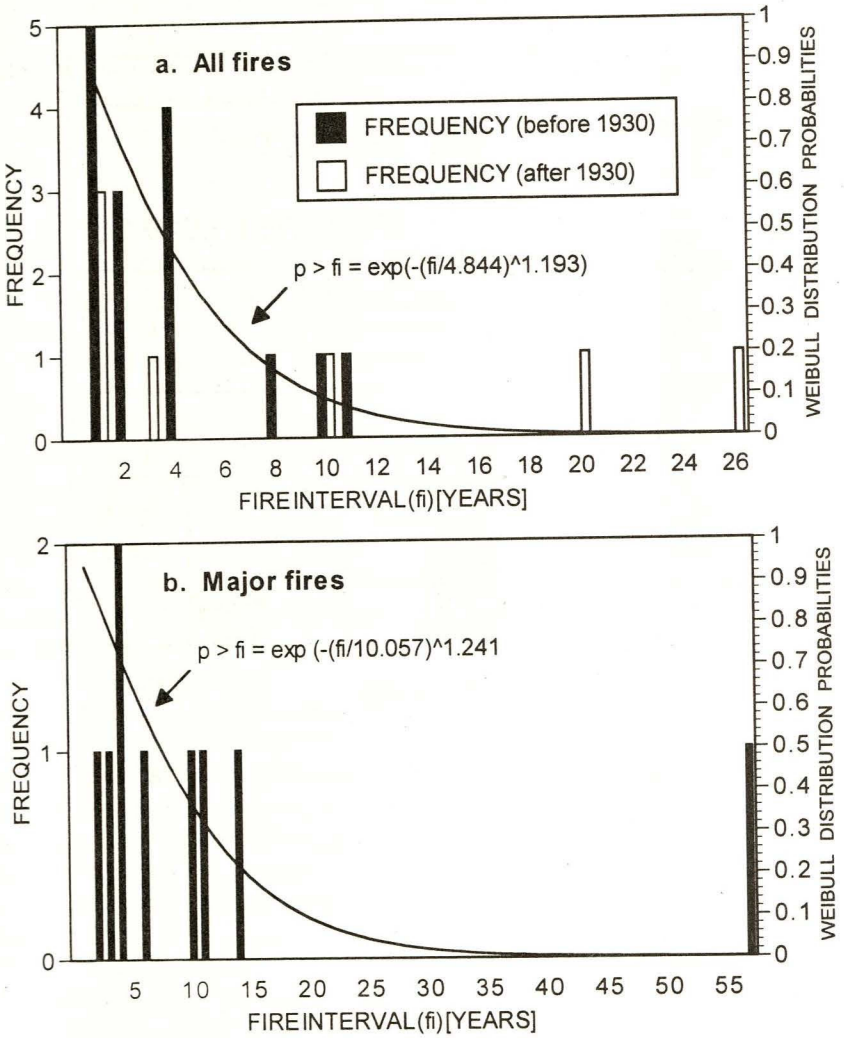


Figure 2. Distribution of fire frequencies, with accompanying Weibull probability distributions. Scale and shape parameters are shown in the equations, where p > fi = exp(-(fi/b)^c), b = scale parameter, c = shape parameter; (a) frequencies for all fires, before and after the fire suppression era (1930); (b) frequencies for major fires (25% of samples are scarred, or at least two samples). The only major fire after the fire suppression era is the longest interval (57 years).

Table 2. Fire intervals (years) at Oreton, Vinton County, Ohio 1856 to 1995. Data are Weibull, not normally distributed. The Weibull intervals indicate significantly ($p < 0.05$) short or long intervals between fires. Major fires are defined as when ≥ 25% of all samples scarred or at least two samples scarred in a given year.

| | All fires | Major fires |
|---|---|---|
| Weibull median interval | 3.6 | 7.5 |
| Weibull short interval | 0.4 | 0.9 |
| Weibull long interval | 12.2 | 24.3 |
| Mean fire interval (MFI) | 5.4 | 11.3 |

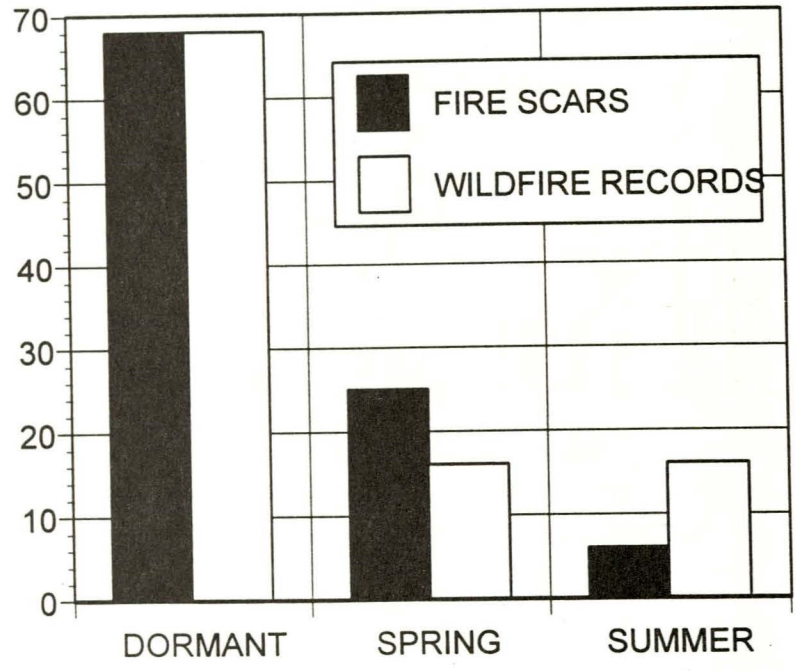


Figure 3. Distribution of fire frequencies by season, comparing fire scar records to local wildfire records.

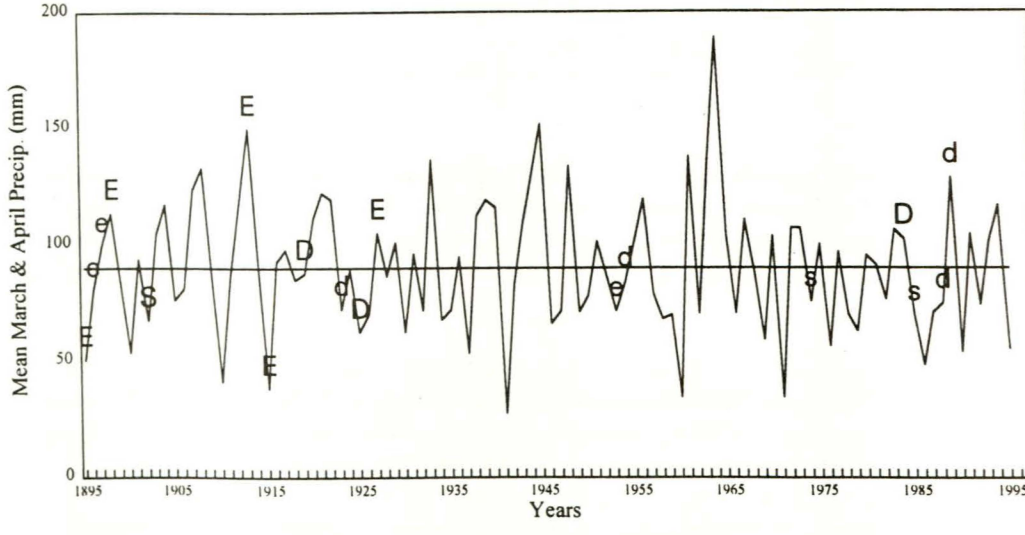


Figure 4. Climate and fire relationships Oreton, Vinton County, Ohio, 1895 to 1995. Mean March and April precipitation (mm) is plotted each year, and fire occurrences are labeled. Precipitation data are for the Southeast Ohio division (National Climatic Data Center, 1996). The horizontal line indicates mean regional March and April precipitation. The labels indicate season when the fire burned (D = dormant season, from August to mid-April but probably late March to mid-April; E = early spring, from mid- to late-April; and S =summer, early May to late July. ) Uppercase letters indicate a major fire; lowercase letters indicate a minor fire.

## DISCUSSION

This even-aged stand was about 140 years old, an age that is consistent with known land use patterns in the Hill Country, where the forests used for charcoal production, agriculture, and pasturage were abandoned and then naturally reforested in the mid to late 19th century.

### Fire Regime

Type of fire. The eastern deciduous forest has predominantly low-intensity surface fires carried by hardwood leaf litter (Fuel Model 9, Anderson 1982). The overstory at Oreton is even-aged, which in other situations might be evidence of a severe, stand-initiating event such as catastrophic fire. However, given the area's land-use history, regrowth after intensive cutting is more likely than replacement after catastrophic fire.

Seasonality. Most fires occurred when the tree cambium was dormant, from about August 1 to April 14. Thus, the season of dormant fires is broadly defined, though wildfire records indicate that spring is the most important fire season, and most dormant scars probably occurred during early March to mid-April: a third of the fires might have happened during autumn. Haines and others (1975) found that the week of April 1 was the peak of the Wayne National Forest spring fire season. The timing of fires during earlywood formation probably can be narrowed to a short season in the spring, and fires during latewood formation probably occurred in late spring or early summer. Summer fires were rare because in these closed-canopy forests, leafy canopies reduce solar radiation at the forest floor while fuel moisture remains high (Haines and others 1975).

Frequency. Virtually all (99%) of fires in the Ohio River Valley are caused by people (arson or accidental ignition); few are caused by lightning (Haines and others 1975, Yaussy and Sutherland 1994). In vegetation types where people are the most important source of ignition, fire frequency reflects occupancy and culture (Whelan 1995). The more people there are, the more ignitions occur (Whelan 1995). In the last century, human populations in the Midwest have increased but fire suppression has been applied aggressively. For example, in the Ozark region of Missouri Guyette and Cutter (1991) developed a fire history using *Q. stellata* (post oak) trees. They reported reduced fire frequency after abandonment of the area by Native Americans, increased frequency after initial colonization by Euro-Americans, and decreased fire frequency during the 20th century era of fire suppression.

Between 1871 and 1989, fire occurred repeatedly at the Oreton site. Small or low-intensity fires that scarred few trees were likely (Weibull median probability interval) every 3 to 4 years, and widespread fires that scarred at least 25% of the sampled trees were likely every 7 to 8 years. In fact, there are only three decades (1930's, 1940's, and 1960's) without fire. There are several notably long intervals between fires in the Oreton record. In Figure 2a, both long intervals occur in the fire-suppression era (i.e., after 1930). Similarly, in the major fire distribution, the longest interval is after 1930. These intervals are outliers from the predicted Weibull probability distributions, where short-interval fires are most probable. Thus, long intervals between fires in this mixed-oak forest are unusual and probably an artifact of fire-suppression policy.

The mean fire intervals at the Oreton site for all fire events (5.4 years) and for major fires (11.3 years) are similar to those observed from other oak-dominated forests in the Midwest. (Note that other studies in similar forest types used the empirical descriptor of frequency, the MFI, and not the Weibull median probability interval; presumably, the MPI's would have been shorter. In a post oak forest in southern Missouri, Guyette and Cutter (1991) noted an MFI (1810 to 1989) of 6.4 years for all events and 11 years for severe events (defined as three or more trees scarred). Similarly, in an oak-hickory/glade transition area in the Ozarks of Missouri, Guyette and McGinnes (1982) noted MFI's of 3.2 years for all events, 8.75 years for two or more trees scarred, and 20 years for three or more trees scarred (1730 to 1870) from scars on *Juniperus virginiana* (eastern redcedar) trees. Alternatively, Cutter and Guyette (1994) noted in post oak stands in south-central Missouri, fire frequency decreased to 24 years during the postsettlement period (circa 1850 to present) from an MFI of 2.8 years before settlement (1740 to 1850). Henderson (1982) calculated fire frequencies of 5.2 years (1933-1981) and 11.1 years (1929-1981) in two neighboring black oak stands in northwestern Indiana. Abrams (1985) found MFI's ranging from 11.2 to 19.7 years in three oak gallery forests in tallgrass prairie in Kansas. Because Abrams used the presence of charcoal in the wound as a

criterion for denoting a scar as fire-caused and because of his reported experimental evidence that not all fires caused scars, the actual fire interval probably is shorter than the estimated interval.

More fire events probably occurred at Oreton than were recorded by scars, and the estimated frequencies likely are conservative for several reasons. First, not all trees at the site were sampled, so some scarring events might have been missed. Second, not all fires cause scars (Abrams 1985). The higher the fire intensity, the higher the probability of scarring, so low-intensity fires are less likely to cause scarring (Guyette and Cutter 1991). Third, most of the trees sampled here had small fire-caused wounds that healed rapidly, so there were no consistent fire-susceptible faces. For example, on open scar faces on *Q. douglassii* H. & A. (blue oak) in California, Mc Claran (1988) found that fire scars were more frequent than in embedded scars in intact trees. Fourth, as the trees grew larger, the bark thickened. With thickening bark and without a fire-susceptible face, trees probably became less susceptible to scarring with time, and fires in later years might have been less likely to cause a scar than earlier fires. Conversely, it is possible that some scars were not caused by fire; as mentioned previously, other factors can cause scarring (Arno and Sneck 1977, Mitchell and others 1983). Scars caused by other sources would lead to overestimates of fire frequency.

Spatial extent and intensity. Some fires scarred many trees, such as the 1913 fire that scarred seven trees, but other fires like the 1923 fire scarred only one tree (Figure 1, Table 1). There are four potential reasons for these differences in number of trees scarred, including variation in (1) fire intensity, (2) fuel quantity and continuity, and (3) spatial extent of fires (due to fire suppression efforts) and 4) decreasing susceptibility to scarring over time (same-aged trees growing increasingly thicker bark). The lower the intensity of fires, the fewer trees are likely to be scarred. It also is likely that the reverse is true.

Fire behavior and hence intensity is a function of slope, fuel, and weather (Rothermel 1980). Within this site, fuel and weather varied in time. Results from this study did not show a relationship between seasonal climatic variables and major or minor fire events. If fire intensity is not driven by climate, fuel loading might explain spatial extent and fire intensity patterns; the longer the interval between fires, the greater the buildup of fuel (to the point where the rate of decay is equal to the rate of buildup). Before the era of fire suppression, fires likely burned freely until they were extinguished by precipitation or encountered a barrier. Additionally, long fire intervals probably ended in relatively intense fires that scarred more trees. But during the fire suppression era, fires were suppressed promptly, minimizing fire size and resulting in little or no scarring. Longer intervals also might reflect increasing resistance to fire-scarring in the increasingly older, thick-barked trees.

## CONCLUSIONS

Many low-intensity fires, both large and small, occurred at this site in southern Ohio at frequencies similar to those for other Midwestern oak forests. The first fire was recorded in the 1870's, and despite fire-suppression efforts after 1930, fire has occurred in 10 of the 13 decades since the first fire. Most fires occurred during the dormant season, probably in the early spring (March-April). There is little indication that climate patterns are driving fire events at this site. Since nearly all ignitions are human-caused, frequency probably is a function of human population density and cultural factors (fire suppression), and the period of time required to develop a continuous layer of leaf litter, the fine fuel that carries fire in deciduous forests.

## ACKNOWLEDGMENTS

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# VEGETATION-SITE RELATIONSHIPS AND FIRE HISTORY OF A SAVANNA-GLADE-WOODLAND MOSAIC IN THE OZARKS

Sean E. Jenkins[1], Richard Guyette[2], and Alan J. Rebertus[3]

Abstract: There is a growing interest in reconstructing past disturbance regimes and how they influenced plant composition, structure and landscape pattern. Such information is useful to resource managers for determining the effects of fire suppression on vegetation or tailoring prescribed fires to restore community and landscape diversity. In the spring of 1995, the National Park Service reintroduced landscape-scale prescribed fire to an extensive oak/pine woodland-savanna-glade complex on Turkey Mountain, Buffalo National River, Arkansas. We took advantage of this opportunity to: (1) reconstruct the fire history of Turkey Mountain from fire scars and (2) determine how plant composition, forest structure, and past fire intensity varied along topoedaphic gradients before the site was burned. A fire history was derived from wedges or cross-sections of one dead and eight live shortleaf pines. Woody and herbaceous vegetation were sampled in 18, 20 by 25-m plots systematically spaced along transects running upslope. The percentage of trees scarred was determined in 80, 20 by 20- or 20 by 40-m plots located along transects running upslope. Highest plant diversity occurred on shallow, calcareous soils (limestone glades). Lowest diversity was associated with deeper, acidic soils and high woody basal area. Vegetation composition changed gradually along environmental gradients, but the most distinct flora was associated with: (1) limestone glades, (2) deeper acid soils, (3) high basal area sites, and (4) sandstone glades. Savanna herbaceous species were characteristic of sites with intermediate soil depth and fertility. Black hickory, post oak, eastern red cedar, and chinquapin oak were overstory dominants. Encroachment of cedar (on circumneutral soils), black jack oak, and black hickory (on acidic soils) may pose a serious threat to the diverse flora. For the 223-year fire record, the mean return interval was 5.7 years. The longest fire-free interval was 34 years (1770-1804) and the shortest was 1 year. No fire scars were documented between 1972 and 1993. The number of trees scarred per plot increased with fetch (distance from the bottom of the mountain), proximity to south-southwest slope aspects (especially on steeper slopes), and distance from the Buffalo River. Past variation in fire frequency and intensity may have acted synergistically with topoedaphic gradients to maintain a mosaic of plant communities.

## INTRODUCTION

Oak ecosystems in Eastern North America have evolved under the influence of fire for thousands of years (Abrams 1992). Savannas, glades, and woodlands in the Ozarks remained open in the past due to fire but were distributed in complex mosaics subtly integrated along continua relating to soil depth and bedrock type. In the Ozarks, glades are treeless or sparsely wooded openings in woodlands, with bedrock at or near the surface (Logan 1992). Glades are often concentrated on, but not limited to, south and west aspects of hillsides, with savanna above and below. Savannas and woodlands are usually defined physiognomically, with the former having less canopy cover (10 to 50%) and more herbaceous groundcover (especially graminoids, composites, and legumes). Savannas and glades are also strongly influenced by the frequency, intensity, and seasonal variation in the occurrence of fire (Ladd 1991).

[1] Ecologist, Biological Research Division, U.S.G.S, Central Plains/Ozark Plateau Field Station, School of Natural Resources, Dept. of Soil and Atmospheric Sciences, 204 Gentry Hall, University of Missouri, Columbia, MO 65211.

[2] Assistant Research Professor, Department of Forestry, School of Natural Resources, University of Missouri, Columbia, MO 65211.

[3] Assistant Professor, Department of Forestry, School of Natural Resources, University of Missouri, Columbia, MO 65211

Since implementation of fire suppression in the Ozarks and other parts of the Midwest, savannas and glades have undergone rapid succession to more closed conditions (Anderson and Schwegman 1991, Apfelbaum and Haney 1990, Beilmann and Brenner 1951, Guyette and McGinnes 1982, Kucera and Martin 1957, and many others). Ozark savannas have become overgrown with black hickory (*Carya texana* Buckl.), black jack oak (*Quercus marilandica* Muenchh.), and a variety of fire-sensitive hardwoods (e.g., shadbush (*Amelanchier arborea* (Michx. f.) Fern.)). Glades have become fragmented by invasion of eastern red cedar (*Juniperus virginiana* L.), black jack oak, winged elm (*Ulmus alata* Michx.), and winged sumac (*Rhus copallina* L.) (Reiter 1991). Fire, cutting, and herbicide application are often recommended to reduce woody density and promote herbaceous diversity, but there is often no explicit rationale for where and how intensively these restoration techniques should be applied across the landscape.

Savannas and glades contain some of the richest floras in North America. For instance, Ver Hoef and others (1991) recorded 365 glade species in the vicinity of the Current River, Missouri, and over 300 species were found in one small restored savanna in central Missouri (Ken McCarty, Missouri Department of Natural Resources, personal communication). Several endangered species, such as Mead's Milkweed (*Asclepias meadii* Torr.), have been recorded in Ozark savannas (Tim Nigh, Missouri Department of Conservation, personal communication). Savannas or "barrens" have widely scattered trees and extremely diverse groundcover vegetation that combines the floristic characteristics of both grasslands and woodlands (Nuzzo 1986). However, research seldom targets transitional areas, and therefore many species remain undetected. Furthermore, glades and savannas provide some of the most striking patch diversity across the Ozark landscape, and the distribution of these communities modifies disturbance regimes, nutrient dynamics, and habitat use of many wildlife species. These communities often represent islands of high diversity (herbaceous plants, insects, and other groups) in a matrix of low diversity, even-aged, closed-canopy forest. Yet, management activities continue to homogenize the landscape through fire suppression and logging without regard to the underlying vegetation patterns.

In the Spring of 1995, the National Park Service reintroduced landscape-scale prescribed fire to an extensive oak/pine woodland-savanna-glade complex on Turkey Mountain, Buffalo National River, Arkansas. The present study was undertaken to provide a better understanding of both the presettlement fire regime and the effects of fire suppression on vegetation. The objectives of this study were to: (1) reconstruct the fire history of Turkey Mountain from fire scars and (2) determine trends in plant composition and past fire intensity along topoedaphic gradients. We were particularly interested in understanding the more subtle variation in vegetation composition across several community types, and how vegetation and fire regimes vary along the same topoedaphic gradients.

## METHODS

### Study Area

Turkey Mountain is located at the junction of the Buffalo and White rivers, Hathaway Wilderness Area, Buffalo National River. A several hundred-hectare oak/shortleaf pine (*Pinus echinata* Mill.) woodland-savanna-glade complex covers the south aspect of Turkey Mountain and north aspect of Granite Mountain. The stratigraphy is complex, consisting of various layers of Ordovician limestone and sandstone. The sandstone "caps" of Turkey Mountain appear to be part of the massive St. Peter formation (Craig 1988, Craig and Deliz 1988). The majority of limestone glades on mid-slopes probably are found on dolomitic limestones of the Powell and Cotter formations. Topography is very steep, with 100-150 m of local relief.

### Vegetative Sampling

Eighteen 500-$m^2$ (20 by 25-m) plots were established in 1993 and permanently marked with rebar on Turkey Mountain and adjoining Granite Mountain. Plots were located 100 m apart along 5 transects running upslope. Transects were subjectively located to cover representative vegetation assemblages, soil types, and slope aspects within the proposed burn unit (Table 1). Herbaceous vegetation was surveyed in June and again in early September. Some pre-vernal species may have already become senescent by late June, but most species were probably detected in these two periods. Cover class values between 0 and 5 (20% cover-scale increments) were assigned to all herbaceous and woody ground cover species (<1 m tall) in each plot. In addition, all trees ≥ 5 cm dbh (diameter at

**Table 1. Site and soil characteristics of Turkey Mountain (TM) and Granite mountain (GM), Buffalo National River, Arkansas**

| Plot | Slope Pos. | Substrate | Dominant Trees/Community | Elev. (m) | Aspect (°) | Slope (°) | Moisture Index 0 = dry | Soil Depth (cm±SD) | Salt pH | CEC (meq/100g) |
|---|---|---|---|---|---|---|---|---|---|---|
| 1 (TM) | upper | chert/sandstone | black oak-black hickory | 311 | 150 SE | 22 | 19 | 5.90 ± 2.6 | 5.10 | 10.45 |
| 2 (TM) | mid | limestone | blackjack oak-black hickory | 253 | 176 S | 26 | 15 | 14.10 ± 9.3 | 5.00 | 5.90 |
| 3 (TM) | lower | limestone | post oak, gladey | 201 | 179 S | 18 | 24 | 10.80 ± 8.3 | 6.75 | 14.65 |
| 4 (TM) | upper | limestone | chinquapin oak, gladey | 262 | 170 S | 20 | 21 | 2.90 ± 4.6 | 7.30 | 11.45 |
| 5 (TM) | mid | limestone | chinquapin oak, gladey | 213 | 175 S | 19 | 19 | 7.90 ± 7.5 | 6.90 | 13.00 |
| 6 (TM) | lower | limestone | eastern red cedar, gladey | 184 | 185 S | 19 | 26 | 4.40 ± 3.7 | 6.85 | 22.65 |
| 7 (TM) | ridge | sandstone | black hickory-blackjack oak, gladey | 275 | 115 E | 8 | 20 | 30.40 ± 24.9 | 5.00 | 8.40 |
| 8 (TM) | ridge | sandstone | black hickory | 301 | 82 E | 8 | 23 | 13.60 ± 7.6 | 6.05 | 36.25 |
| 9 (TM) | ridge | sandstone | blackjack oak-post oak, gladey | 283 | 230 SW | 4 | 11 | 18.40 ± 12.8 | 4.85 | 8.70 |
| 10 (TM) | saddle | limestone/sandstone colluvium | black hickory-chinquapin oak, open glade | 267 | 115 E | 8 | 23 | 5.00 ± 6.5 | 6.60 | 10.10 |
| 11 (TM) | lower | limestone | eastern red cedar, gladey | 173 | 125 ESE | 11 | 36 | 15.90 ± 16.1 | 6.75 | 16.50 |
| 12 (TM) | upper | limestone | post oak-northern red oak | 250 | 290 WNW | 21 | 23 | 8.30 ± 8.3 | 6.45 | 28.35 |
| 13 (GM) | upper | limestone | post oak | 263 | 242 WSW | 20 | 15 | 4.20 ± 3.8 | 6.45 | 15.40 |
| 14 (GM) | lower | limestone | black hickory-eastern red cedar | 197 | 20 NNE | 12 | 46 | 13.40 ± 16.1 | 6.55 | 18.15 |
| 15 (GM) | mid | limestone | eastern red cedar, gladey | 218 | 325 NW | 12 | 32 | 9.80 ± 12.2 | 7.15 | 32.10 |
| 16 (GM) | upper | limestone | chinquapin oak, gladey | 251 | 331 NW | 18 | 25 | 5.40 ± 5.4 | 6.85 | 19.30 |
| 17 (GM) | ridge | chert/sandstone | white oak-southern red oak | 288 | 349 NW | 4 | 30 | 40.10 ± 9.7 | 4.60 | 3.10 |
| 18 (TM) | lower | chert | shortleaf pine | 155 | 163 SSE | 24 | 26 | 22.10 ± 7.8 | 4.45 | 6.90 |

breast height) were tallied and dbhs were recorded. Tree cores were taken at 0.4 m aboveground from about 10 trees in each plot. Trees were subjectively chosen to represent the range of sizes for each species and past periods of recruitment. Saplings (2.5 ≤ dbh < 5 cm), large seedlings (>0.5 m tall; < 2.5 cm dbh), and small seedlings (<0.5 m tall) were tallied in 100-$m^2$, 10-$m^2$, and 1-$m^2$ circular subplots, respectively. Two subplots of each size were located in each main plot. The total area sampled for small seedlings, 36 $m^2$, was insufficient to provide good density estimates for most species; thus, these data should be interpreted with caution.

Soil depths were measured with a steel rod at 16 points located at 4-m intervals along the plot diagonals. Soil samples were taken from the upper 15 cm from the center of each quarter of the plot. The two upper and two lower quarters were pooled separately. Samples were frozen for several months, sieved, and analyzed by the Soil Characterization Lab at the University of Missouri. In general, the within-plot variation in soil properties was very small, so only plot averages are presented in the results. Elevation, slope aspect, slope steepness, topographic position, and slope shape were recorded at each plot and used to compute a topographic relative moisture index (TRMI; Parker 1982). The TRMI is lowest for steep, upper, convex, southwest-facing slopes, and highest for valley bottoms or lower, northeast-facing slopes.

Fire History

Tree-ring analysis of fire scar dates was used to document the fire history of Turkey Mountain. Wedges or cross-sections were cut from eight living shortleaf pines and one well-preserved shortleaf pine snag. The snag, located on an upper slope, had 31 fire scars and recorded most of the fire years. Scars were cross-dated by comparison with cores from Turkey Mountain and chronologies from Leatherwood Creek and the Current River and Piney River watersheds in Missouri.

We examined variation in past fire intensity along topographic gradients by counting the number of trees scarred in 80, 20 by 20-m plots, systematically spaced every 15 m in vertical relief along eight transects running from the base to the summit of Turkey Mountain. Where trees were sparse (e.g., near glades), 20 by 40-m plots were used. As a general rule, only trees ≥ 15 cm dbh were included. At each plot we also recorded aspect, slope, elevation (by altimeter, 0.9-m resolution), estimated range of tree diameters, and tree species. Fetch (vertical relief from the bottom of the mountain) and the distance of each plot to the Buffalo River were determined from a USGS topographic map. We suspected that the Buffalo River would be a major anthropogenic ignition source, so we measured distance to the point on the river where Cook Creek enters, following the most likely path that fire would burn into the area.

Numerical Analyses

Multiple regression analysis (PROC GLM; SAS 1988) was used to model the percentage of trees scarred with respect to aspect, slope steepness, fetch, distance to the Buffalo River, and interactions among these variables. We transformed aspect, a circular variable, to a linear scale by treating all azimuths as deviations from 205°, approximately the most xeric aspect in terms of solar radiation and moisture demand (Parker 1982).

We used ordination by Canonical Correspondence Analysis (CCA) to examine trends in vegetation composition along environmental gradients. Ordination objectively arranges plots and (or) species together based on their degree of similarity. CCA uses measured environmental variables to constrain plots and species ordinations. The computer program CANOCO (Ter Braak 1988) was used for all analyses. Untransformed cover values were used for groundcover vegetation. Plot importance values (relative density + relative basal area + relative frequency /3) were used for trees ≥ 5 cm dbh. A square root transformation was used for tree data to reduce the effects of dominant species. For the understory stratum, we used relative densities that were weighted by size class: 1 for saplings, 0.1 for large seedlings, and 0.01 for seedlings. Rare species were down-weighted and scaling was symmetric; otherwise, all defaults in CCA were used. Environmental variables used in the ordination included: elevation, a topographic moisture index or TRMI, salt (0.01 M $CaCl_2$) pH, cation exchange capacity (CEC), total bases, percent organic matter, percent sand, total plot basal area, importance values of major overstory species, and the predicted percent trees scarred. Soil, topographic, and overstory variables were initially screened separately; then, "forward selection" techniques within CANOCO were used to identify a final subset of variables that were most correlated

with the vegetation data sets. Monte Carlo permutation tests were used to determine the significance of the first canonical axis for overall significance of the species-environmental relationship.

## RESULTS

### Fire Regime

We cross-dated 54 scars on the 9 pines, and for the 223 years of record the mean fire interval was 5.7 years (Figure 1). The longest fire-free interval was 34 years (1770-1804) and the shortest was 1 year. During the 1950s, Turkey Mountain burned almost annually. Only one fire year, 1953, was found on more then one tree during that decade. No fire scars were documented between 1972 and 1993. Our fire history provides only minimum estimates for return intervals; the small sample size probably underestimated the frequency of fire on Turkey Mountain.

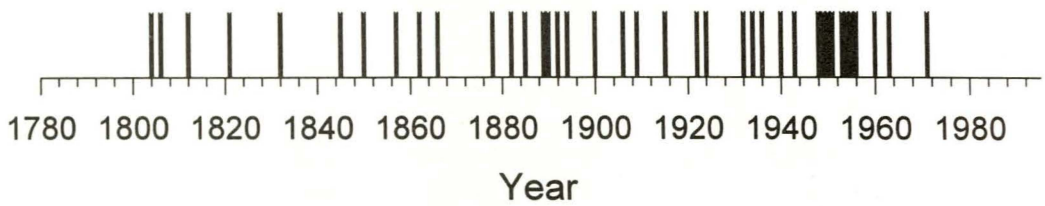


Figure 1. Fire-year chronology for Turkey Mountain, Arkansas.

The percentage of trees scarred (not dated) in the plots was significantly related to fetch (FETCH, in feet), the interaction of slope steepness and aspect (*SLP x ASP*), and distance from the Buffalo River (DIST, in miles):

$$\%\ scarred = -0.58+(0.098*FETCH)+(0.0069*SLP\ x\ ASP)+(11.65*DIST);\ r^2=0.75,\ F^{3,76}=45.45,\ P<0.001.$$

The percent of tree scarred increased with fetch and on steeper south- and west-facing slopes (Figure 2). The mean percent of trees scarred ranked by aspect were as follows: south (52%), west (47%), east (43%), and north (37%). The interaction of slope steepness and aspect contributed more to the regression model than either variable alone: the percent of trees scarred increased with greater slope steepness on south and west aspects, but steepness had little effect on north and east aspects (Figure 2).

### Age Structure

The age structures of hardwood tree species on Turkey Mountain were distinctly multimodal, with even-aged cohorts establishing in the 1910s, 1940s and 1960s (Figure 3). The oldest tree was a 188-year-old post oak (*Quercus stellata* Wang.), but most of the large trees were hollow and could not be dated. The majority of eastern red cedar and shortleaf pine trees on the lower slopes established c. 1925-1940, a period when the mountain was burned relatively frequently (see Figure 1), although the extent of the burns is unknown. It should be cautioned that this was not a random sample of tree ages and that coverage was fairly limited. Also, <50% of the cores were sound enough to estimate complete ages.

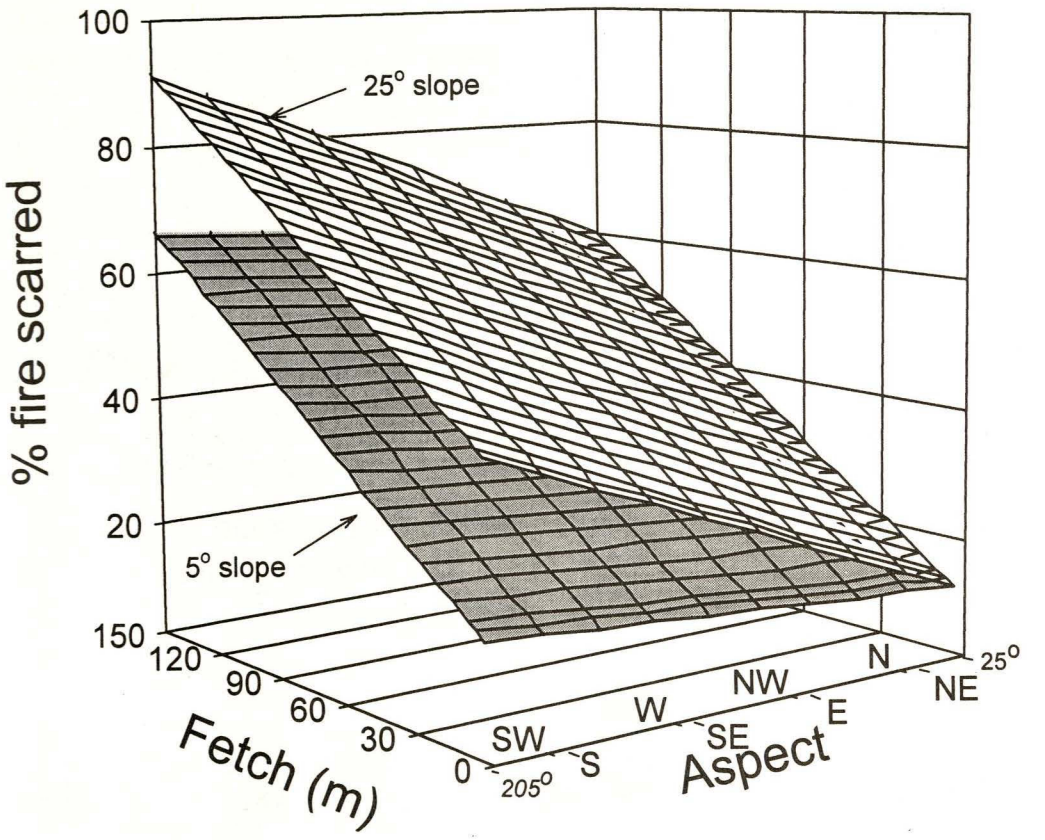


Figure 2. Percentage of trees scarred as a function of fetch (distance from bottom of mountain), aspect, and slope steepness. Aspect is given as degrees deviation from 205° azimuth. Two levels of slope steepness are shown.

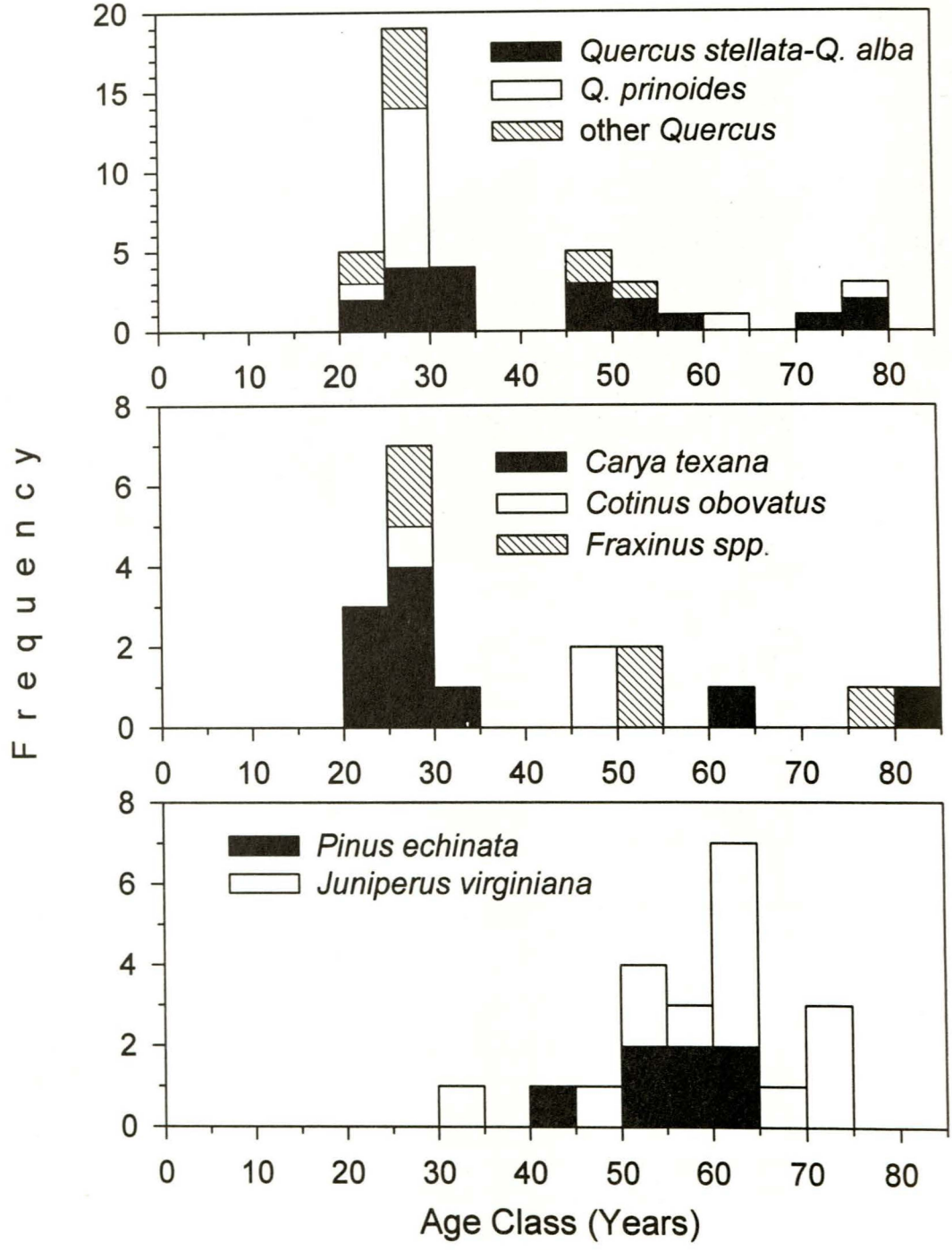


Figure 3. Age distribution of trees on Turkey Mountain. Trees > the 85 years are not shown.

Vegetation

There were 26 tree and tall shrub species (≥ 5 cm dbh) found on Turkey Mountain, with black hickory, eastern red cedar, post oak, and chinquapin oak (*Quercus prinoides* Willd.) having the highest importance values (Table 1). Tree compositional patterns were strongly related to environmental variables (Figure 4). The first two ordination axes accounted for 36% of the variance in species data and 59% of the variance in the species-environment relation (see Ter Braak 1988). The first canonical axis and overall species-environmental relationship were both highly significant (Monte carlo tests, P=0.01) . Plot 18 was the most distinct, characterized by pine and sassafras on deeper soil with low pH, CEC, and total bases (Figure 4). Plots 1, 2, 7, 9, and 17 were also on deeper, more acid soils overlying sandstone or chert; with indicator species such as black jack oak, black locust (*Robinia pseudo-acacia* L.), southern red oak (*Quercus falcata* Michx.), white oak (*Q. alba* L.), and black oak (*Q. velutina* Lam.). White oak was apparently restricted more by soil depth than pH. The rest of the plots were on shallow soils overlying limestone. Indicator species included typical calcicoles: blue ash (*Fraxinus quadrangulata* Michx.), eastern red cedar, gum bumelia (*Bumelia lanuginosa* (Michx.) Pers.), chinquapin oak, and American smoke tree (*Cotinus obovatus* Raf.). Chinquapin oak importance values were also highly correlated with magnesium levels (not shown on ordination). Post oak, persimmon (*Diospyros virginiana* L.), and northern red oak (*Quercus rubra* L.) were the most widespread, but were slightly more abundant on circumneutral soils.

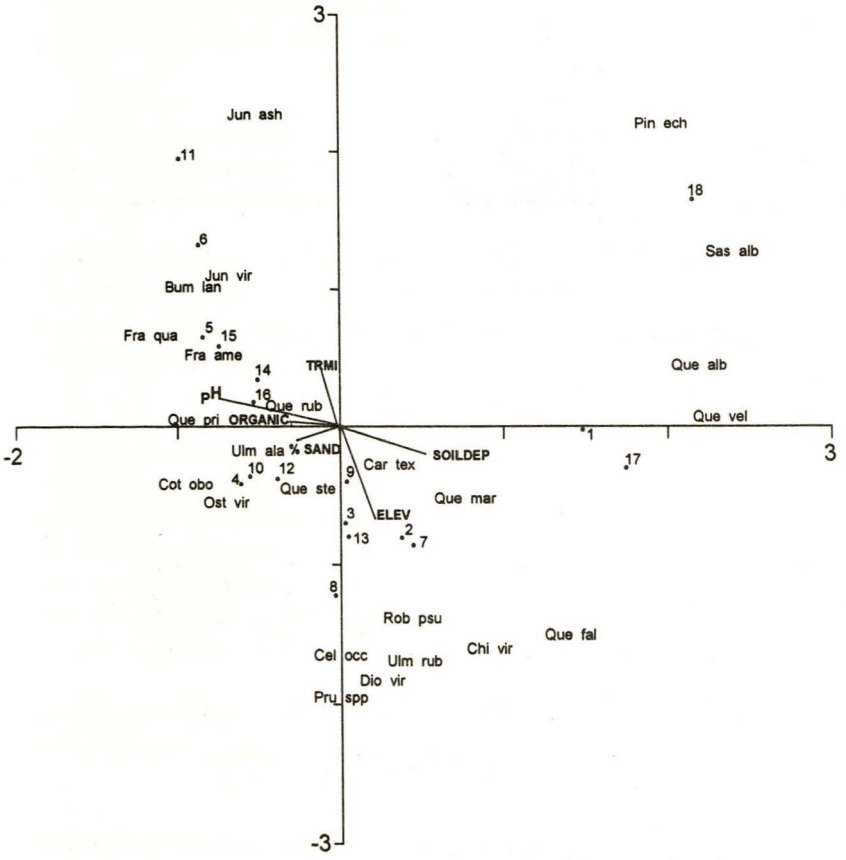


Figure 4. CCA ordination triplot of tree and tall shrub species (≥ 5 cm dbh) on Turkey Mountain. The environmental vectors point in the direction of maximum correlation and vector length is proportional to the strength of the correlation. Numbers represent plots and species codes include the first three letters of the genus and specific epithet (see Appendix 1).

The understory stratum was dominated by black jack oak, black hickory, post oak, and eastern red cedar (Table 1). Black jack oak and black hickory pose the greatest threat to sandstone glade and savanna vegetation; their abundance was clearly correlated with lower groundcover diversity (see Figure 6). Savanna species, such as post oak, are still reasonably abundant in the understory strata. The woody understory vegetation was less strongly related to measured environmental gradients than the overstory (data not shown). The first and second ordination axes accounted for 27% of the variance in species data and 56% of the variance in the species-environment relation. The first canonical axis was significant (P = 0.01), and the overall species-environmental relationship was also significant (P = 0.01). Plots on deeper and/or acid soils were characterized by fringe tree, winged sumac, black oak, white oak, sassafras, and black jack oak. Basic soils were characterized by the same calcicoles as in the overstory plots (see above); as well as winged elm, southern rusty blackhaw (*Viburnum rufidulum* Raf.), and rough-leaf dogwood (*Cornus drummondii* Meyer). Persimmon, post oak, and shadbush displayed a wide range of occurrence.

The composition of ground cover vegetation was highly variable with respect to topoedaphic gradients. The first and second ordination axes only accounted for 23% of the variance in species data and 42% of the variance in the species-environment relation. The first canonical axis was marginally signifcant (P = 0.06), but the overall species-environmental relationship was highly significant (P = 0.01). Vegetation composition tended to change gradually along environmental gradients (Figure 5). Plots 2, 7, 9, and 17—all on acid soils—were characterized by a tick trefoil (*Desmodium pauciflora* (Nutt.) DC.), two sedges (*Carex muhlenbergii* Schk., *C. retroflexa* Muhl.), a nut sedge (*Cyperus filiculmis* Vahl.), several grasses (*Sphenobolis intermedia* Rydb., *Elymus virginicus* L., *Sporobolus asper* (Michx.) Kunth., *Aristida dichotoma* Michx., *Panicum linearifolium* Scribn.), hairy lip fern (*Cheilanthes lanosa* (Michx.) D.C. Eaton), dwarf dandelion (*Krigia dandelion* (L.) Nutt.), purple cudweed (*Gnaphalium purpureum* L.), butterfly pea (*Clitoria mariana* L.), pellitory (*Parietaria pensylvanica* Muhl.), St. Andrew's cross (*Ascryum hypericoides* L.), forked chickweed (*Parinychia fastigata* (Raf.) Fern.), a sunflower (*Helianthus hirsutus* Raf.), a rushfoil (*Crotonopsis linearis* Michx.), and others (Figure 5).

Plots 7 and 9 are partly on sandstone glade (soil depth ranged from 0 cm on the exposed bedrock ledge to 50-80 cm further upslope). Indicator species on these shallow, acidic soils included stonecrop (*Sedum nuttallianum* Raf.), a brome grass (*Bromus racemosus* L.), and many of the same species listed for acidic soils above.

Plots 1, 8, and 18 were associated with high basal area (closed woodland canopy) and soils with low sand (high silt) content. Indicator species included *Vaccinium* spp., summer grape (*Vitis aestivalis* Michx.), and wild yam (*Dioscorea quaternata* (Walt.) J.F. Gmel.); in addition to more shade-tolerant woodland herbs, such as three-lobed violet (*Viola triloba* Schwein.), alum root (*Heuchera americana* L.), and wild bergamot (*Monarda fistulosa* L.) (Figure 5b and c ).

Most of the plots on basic soils grouped very closely together, floristically (plots 3, 4, 5, 6, 10, 11, 12, 13, 14, 16). Characteristic species for these plots included purple prairie clover (*Petalostemon purpureum* (Vent.) Rydb.), false boneset (*Kuhnia eupatorioides* L.), two bluets (*Houstonia nigricans* (Lam.) Fern., *H. tenuifolia* Nuttall.), prairie acacia (*Acacia angustissima* (Mill.), two tick trefoils (*Desmodium sessifolium* (Torr.) T. & G., *D. illinoense* Gray), blue-eyed grass (*Sisyrhinchium campestre* Bickn.), squaw-weed (*Senecio obovatus* Muhl.), a coreopsis (*Coreopsis palmata* Nutt.), side-oats grama (*Bouteloua curitipendula* (Michx.) Torr.), a rock pink (*Talinum calycinum* Engelm.), and dwarf hackberry (*Celtis tenuifolia* Nutt.) (Figure 5b and c).

Physiographically, plot 14 is the most mesic site (lower slope, north aspect) and it had several distinct species: wild yam, agrimony (*Agrimonia rostellata* Wallr.), lyre-leaved sage (*Salvia lyrata* L.), a morning glory species (*Ipomoea* sp. L.), and a crown-beard (*Verbesina helianthoides* Michx.) (Figure 5 b and c).

A total of 251 species (154 genera, 62 families; 202 herbaceous/semi-woody and 49 woody species) were recorded on Turkey Mountain; 240 were recorded in plots (Appendix 1). Species richness was highest in plots on limestone glades (Figure 6). Low diversity was correlated with acid soils and high basal area (closed canopy). Relatively low diversity was also correlated with high importance values of black jack oak (also on acidic soil), where thickets were heavily shaded. The high density of black jack oak stems was probably a result of fire suppression starting in the 1970's after the area became part of the Buffalo National River. High basal area was also correlated with the deeper

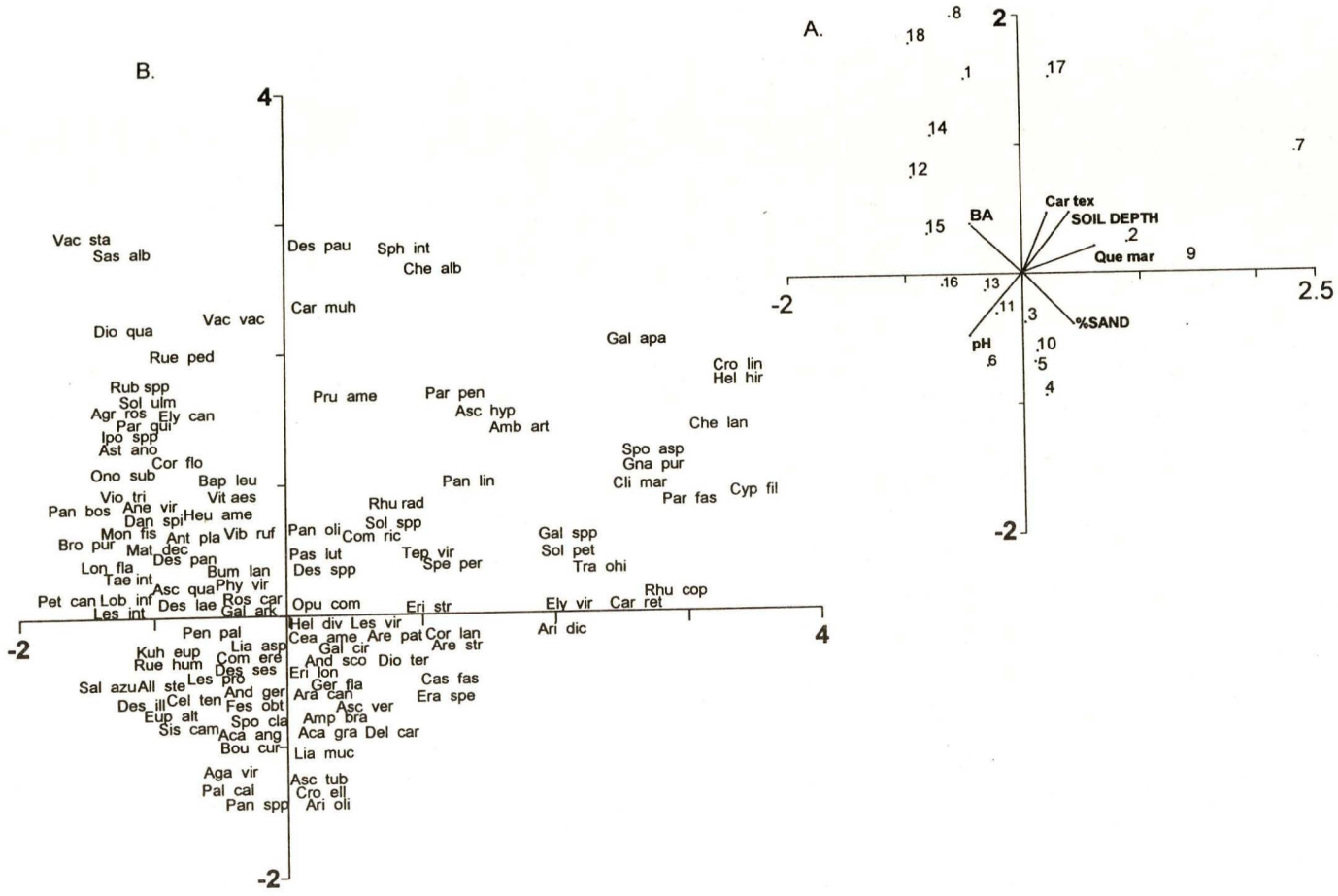


Figure 5. CCA ordination of plots (A) and groundcover species (B and C). Figure 5C shows species locations in the vicinity of the origin, where not all species were labeled in Figure 5B. See Figure 4 for further explanation of ordination diagrams.

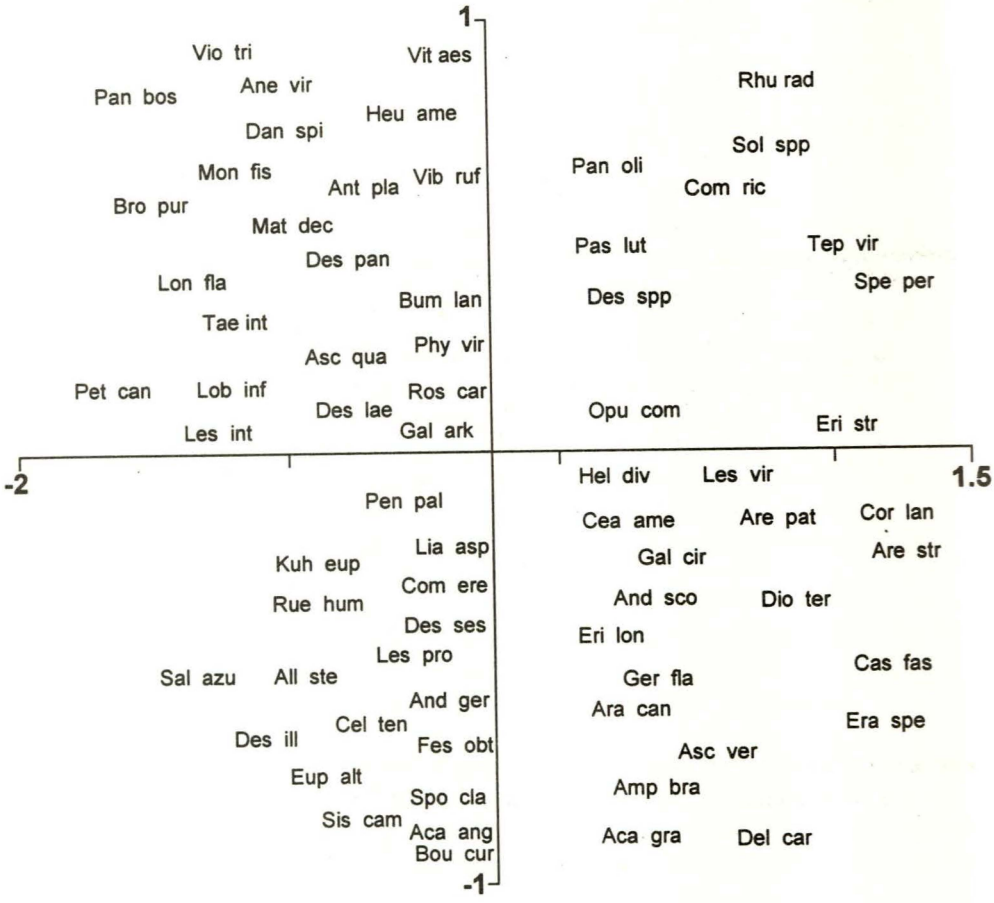
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Figure 5c.

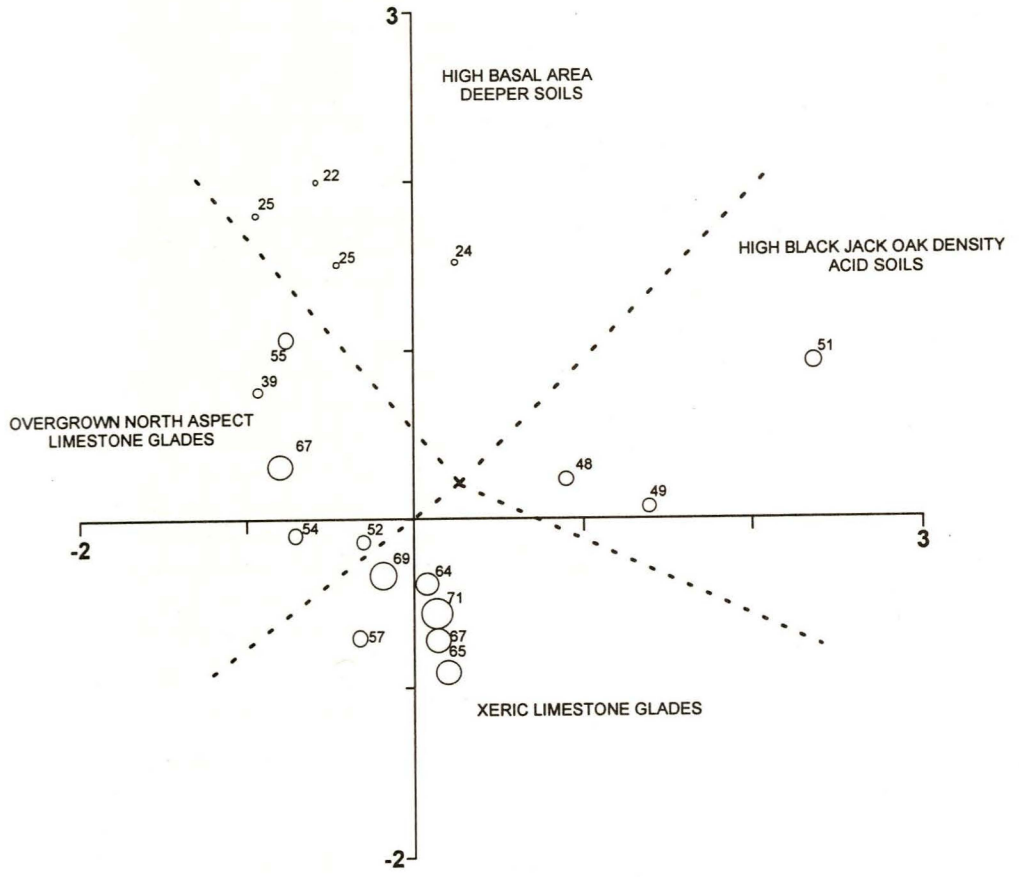


Figure 6. Species richness (number of species) of plots on Turkey Mountain. Plots are positioned in CCA ordination space and the environmental vectors are the same as those in Figure 5a. Circle size is proportional to species richness.

acidic soils located on the upper (shoulder) slopes and summits of knobs (Table 2), where more rapid invasion of black jack oak and black hickories has occurred.

## DISCUSSION

Fire has influenced vegetation in the area for at least the past two centuries. The mean fire return interval, 5.7 years, is very similar to fire frequencies documented for oak and oak-pine sites in the Missouri Ozarks (Guyette and Cutter 1991, Cutter and Guyette 1994). Fire frequency in the Ozarks is highly dependent on ignitions from human populations because of the rarity of lightning caused fires (Guyette and Cutter 1991). The absence of fire scars in the early period, 1770-1804, is weak evidence for the absence of fire in the area because the older part of this record is based on only one tree, and the first fire scar generally requires more heat than subsequent scars (Gutsell and Johnson 1996). Alternatively, 1804 may indicate the beginning of frequent burning by the Cherokee, who migrated into the area during the period 1785-1828 (Pitcaithley 1987). The Cherokee began to leave Arkansas by treaty in 1828. The few fire scars between 1815 and 1840 may indicate a true reduction in fire frequency despite the data being from only one tree. Fire scar data from a post oak savanna 50 km to the north showed a similar reduction in fire frequency between 1810 and 1850 (Guyette and Cutter 1991).

**Table 2. Summary of woody vegetation on Turkey mountain. Sap = saplings, LS = large seedlings, SS= small seedlings, IV = importance value. For overstory trees IV = ((relative + relative basal area + relative frequency/3)*100). For woody regeneration IV = ((relative density + relative frequency/2)*100).**

| | TREES | | | | REGENERATION | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Species | Frequency (% of plots) | Basal Area ($m^2$)/ha | Density (stems/ha) | I.V. | Frequency (% of plots) | Sap/ha | LS/ha | SS/ha | I.V. |
| *Carya texana* | 94.4 | 2.37 | 256 | 16.62 | 77.8 | 267 | 170 | 1700 | 12.1 |
| *Juniperus virginiana* | 44.4 | 3.14 | 239 | 15.28 | 55.6 | 139 | 250 | 3100 | 0.0 |
| *Quercus stellata* | 83.3 | 2.70 | 159 | 14.11 | 50.0 | 211 | 360 | 9400 | 11.1 |
| *Q. prinoides* | 66.7 | 1.83 | 117 | 10.32 | 44.4 | 17 | 250 | 3100 | 4.7 |
| *Q. marilandica* | 44.4 | 0.81 | 141 | 7.75 | 38.9 | 425 | 190 | 1900 | 13.0 |
| *Pinus echinata* | 22.2 | 1.90 | 33 | 5.96 | 5.6 | 3 | 0 | 0 | 0.5 |
| *Quercus rubra* | 55.6 | 0.55 | 46 | 5.16 | 44.4 | 31 | 190 | 800 | 4.4 |
| *Q. velutina* | 16.7 | 0.60 | 66 | 3.88 | 16.7 | 83 | 30 | 300 | 3.1 |
| *Ulmus alata* | 50.0 | 0.34 | 19 | 3.71 | 55.6 | 44 | 170 | 800 | 5.4 |
| *Quercus falcata* | 11.1 | 0.68 | 9 | 2.21 | 0.0 | 0 | 0 | 0 | 0.0 |
| *Cotinus obovatus* | 22.2 | 0.11 | 27 | 2.07 | 38.9 | 33 | 60 | 1900 | 4.0 |
| *Quercus alba* | 11.1 | 0.43 | 20 | 1.99 | 11.1 | 14 | 0 | 0 | 1.1 |
| *Bumelia lanuginosa* | 27.8 | 0.05 | 12 | 1.81 | 27.8 | 8 | 60 | 300 | 2.3 |
| *Fraxinus americana* | 22.2 | 0.07 | 11 | 1.55 | 44.4 | 44 | 170 | 300 | 4.5 |
| *Sassfras albidum* | 16.7 | 0.04 | 14 | 1.29 | 27.8 | 97 | 280 | 2800 | 5.3 |
| *Robinia pseudoacacia* | 16.7 | 0.04 | 13 | 1.27 | 22.2 | 33 | 0 | 0 | 2.3 |
| *Diospyros virginiana* | 11.1 | 0.05 | 14 | 1.04 | 27.8 | 22 | 30 | 0 | 2.5 |
| *Fraxinus* | 5.6 | 0.09 | 9 | 0.71 | 5.6 | 6 | 30 | 300 | 0.7 |
| *Ulmus rubra* | 11.1 | 0.04 | 2 | 0.69 | 0.0 | 0 | 0 | 0 | 0.0 |
| *Juniperus ashei* | 5.6 | 0.12 | 2 | 0.58 | 0.0 | 0 | 0 | 0 | 0.0 |
| *Prunus sp.* | 5.6 | 0.01 | 3 | 0.38 | 16.7 | 6 | 30 | 0 | 1.4 |
| *Chionanthus* | 5.6 | 0.01 | 2 | 0.35 | 11.1 | 36 | 110 | 0 | 1.8 |
| *Celtis occidentalis* | 5.6 | 0.01 | 1 | 0.33 | 5.6 | 3 | 0 | 10000 | 2.7 |
| *Rhus copallina* | 5.6 | 0.003 | 1 | 0.31 | 5.6 | 0 | 110 | 0 | 0.6 |
| *Acer saccharum* | 5.6 | 0.002 | 1 | 0.31 | 0.0 | 0 | 0 | 0 | 0.0 |
| *Ostrya virginiana* | 5.6 | 0.002 | 1 | 0.31 | 0.0 | 0 | 0 | 0 | 0.0 |
| *Amelanchier arborea* | 0.0 | 0.00 | 0 | 0.00 | 16.7 | 6 | 60 | 600 | 1.6 |
| *Cornus drummondii* | 0.0 | 0.00 | 0 | 0.00 | 5.6 | 3 | 140 | 0 | 0.8 |
| *Viburnum rufidulum* | 0.0 | 0.00 | 0 | 0.00 | 11.1 | 19 | 80 | 0 | 1.4 |
| *Carpinus caroliniana* | 0.0 | 0.00 | 0 | 0.00 | 5.6 | 11 | 0 | 0 | 0.6 |
| *Cornus florida* | 0.0 | 0.00 | 0 | 0.00 | 5.6 | 3 | 0 | 0 | 0.5 |
| *Crataegus sp.* | 0.0 | 0.00 | 0 | 0.00 | 5.6 | 8 | 0 | 0 | 0.6 |
| *Rhus glabra* | 0.0 | 0.00 | 0 | 0.00 | 5.6 | 3 | 0 | 0 | 0.5 |
| *Vaccinium spp* | 0.0 | 0.00 | 0 | 0.00 | 5.6 | 3 | 110 | 300 | 0.7 |
| *Celtis tenuifolia* | 0.0 | 0.00 | 0 | 0.00 | 16.7 | 3 | 0 | 0 | 1.5 |

The age structures of overstory trees appear to reflect recent variation in fire history. A major even-aged cohort 20-35 years old, for example, corresponds to the cessation of frequent burning; in particular, dating back to extensive burns in 1960 and 1963. Multimodal age structures of oak savannas have been described by Apfelbaum and Haney (1990) in the Upper Midwest and by Jenkins and Rebertus (1994) in the Missouri Ozarks. Distinct even-aged cohorts are thought to originate during fire-free periods that allow stems to grow tall enough to survive the next fire. Jenkins and Rebertus (1994) also found that near-annual burning reduced fuel loads and resulted in very mild, patchy fires that encouraged sporadic regeneration of oaks. Livestock grazing, which was once pervasive in the Ozarks, could also cause very patchy, mild burns by trampling and reducing fine matrix fuels. The establishment of a cedar cohort 50-65 years ago, a period of frequent burning, may illustrate the combined effect of frequent burning and grazing in reducing fuel loads and exposing bare mineral soil for conifer seed germination.

The distribution of fire-scarred trees aptly illustrates the influence of landscape features on fire intensity. Fetch is important because fires tend to burn upslope and gain momentum by convective pre-heating of fuels and the generation of thermal winds (Pyne 1984). Slope aspect and steepness determine the amount of solar radiation received, and also influence temperature, wind exposure, and relative humidity—all factors that influence fire intensity. Steep south- and west-facing slopes at mid to upper slope positions had the highest percentage of trees scarred (Figure 2), and these also tended to be the most xeric sites (Parker 1982). We also found a 22% increase in fire scarring from the mouth of Cook Creek to its upper reaches (about 3.2 km), which may indicate that fires were more frequent, but less intense near a major ignition source (Buffalo River), or simply that fires gained momentum as they travelled upslope from the river. Describing variation in scarring across topographic gradients is potentially confounded by parallel changes in species composition and tree size, two factors known to affect probability of scarring (Gutsell and Johnson 1996). However, the results of our model are consistent with known fire behavior in hilly topography.

One of the major roles fire plays in maintaining species diversity in savannas and grasslands is the control of woody plant invasion (Huston 1994). However, it is difficult to separate the effects of fire regime and topoedaphic gradients on vegetation composition because all these factors are interrelated. For example, the predicted percent of trees scarred in the vegetation plots on Turkey Mountain was highly correlated with elevation ($r=0.83$, $P<0.001$, $n=18$) and the topographic moisture index ($r=-0.77$, $P<0.001$, $n=18$), two variables that also defined a major gradient in woody overstory composition and accompanying fuels (Figure 4). Fertility is also known to influence burning probability: low fertility tends to promote highly pyrogenic, non-woody species (Kellman 1984). With continued fire suppression, there will probably be a continued reduction in herbaceous diversity in the high basal area sites and a slower, but inevitable encroachment of cedar into the very shallow soils on mid and upper-slopes. Continued woody encroachment would result in the continued loss of diversity (reviewed by Heikens and Robertson 1994). In addition, the dramatic changes in light and litter on glades and savannas may result in less distinct flora on these sites. Besides black jack oak and cedar, Reiter (1991) identified winged elm, winged sumac, and aromatic sumac as invaders of glades in southern Missouri. Both winged elm and aromatic sumac are relatively abundant on Turkey Mountain.

The overriding environmental factors controlling the distribution of both woody and herbaceous species on Turkey Mountain are bedrock type; which influences soil pH, CEC, total bases, percent sand, soil depth and topographic position (elevation, TRMI). These variables, in turn, are also interrelated to bedrock stratigraphy. Glades occur on limestone exposed at mid to lower slope positions on both south and west aspects. Chert savanna and sandstone glades are located on upper slope positions and summits. However, subsurface chert and colluvial processes on the steep slopes probably blur some of the floristic types on Turkey Mountain, producing more of a vegetation continuum (see Figure 6). The complex stratigraphy and colluvial processes also increase resource heterogeneity in these generally infertile soils, a situation known to foster high species diversity (Tilman 1982). The high diversity of the limestone glades is probably due to several interacting factors including: (1) low moisture and soil infertility (Tilman 1982), (2) drier, windier conditions conducive to export of herb-smothering litter, (3) periodic disturbances (fire, frost heaving, and erosion) preventing single species dominance (Huston 1994), and (4) slower woody encroachment on shallow soils (Pallardy and others 1991).

Turkey Mountain presents a rare opportunity for using landscape-scale prescribed burns to restore a mosaic of savannas, glades, and woodlands. Topographic variation and its control of fire intensity will reestablish gradients in tree structure and community composition (particularly sun versus shade herbs). Overall species richness remains high at Turkey Mountain because of slow woody encroachment on its shallow soils; however, fire should increase the abundance of many species and promote flowering. Many savanna species can persist under a closed canopy but in a very low state of vigor (McCarty and Hassein 1984). Burning of this landscape not only has historical precedent but will help maintain diversity at multiple scales.

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Appendix 1. Species list and codes for Turkey Mountain. Scientific names follow Steyermark (1955).

| Genus/species | Code | Genus/species | Code | Genus/species | Code |
|---|---|---|---|---|---|
| *Acacia angustissima* (Mill.) | Aca ang | *Carex sp.* L. | Car spp | *Erigeron annus* (L.) Pers. | Eri ann |
| *Acalypha gracilens* Gray | Aca gra | *Carpinus caroliniana* Walt. | Car car | *Erigeron strigosus* Muhl. | Eri str |
| *Acer saccharum* Marsh | Ace sac | *Carya texana* Buckl. | Car tex | *Eriogonum longifolium* Nutt. | Eri lon |
| *Agave virginica* L. | Aga vir | *Cassia fasciculata* Michx. | Cas fas | *Eupatorium altissimum* L. | Eup alt |
| *Agrimonia pubescens* Wallr. | Agr pub | *Ceanothus americanus* L. | Cea ame | *Euphorbia corollata* L. | Eup cor |
| *Agrimonia rostellata* Wallr. | Agr ros | *Celtis occidentalis* L. | Cel occ | *Euphorbia dentata* Michx. | Eup den |
| *Allium stellatum* Fraser | All ste | *Celtis tenuifolia* Nutt. | Cel ten | *Festuca obtusa* Biehler. | Fes obt |
| *Ambrosia artemisiifolia* L. | Amb art | *Cheilanthes lanosa* (Michx.) D.C. Eaton | Che lan | *Festuca octoflora* Walt. | Fes oct |
| *Amelanchier arborea* (Michx. f.) Fern. | Ame arb | *Chenopodium album* L. | Che alb | *Fraxinus americana* L. | Fra ame |
| *Amphicarpa bracteata* (L.) Fern. | Amp bra | *Chionanthus virginica* L. | Chi vir | *Fraxinus quadrangulata* Michx. | Fra qua |
| *Andropogon gerardi* Vitm. | And ger | *Clitoria mariana* L. | Cli mar | *Galactica volubilis* (L.) Britt. | Gal vol |
| *Andropogon scoparius* Michx. | And sco | *Comandra richardsiana* Fern. | Com ric | *Galium aparine* L. | Gal apa |
| *Anemone virginiana* L. | Ane vir | *Commelina erecta* L. | Com ere | *Galium arkansanum* Gray | Gal ark |
| *Antennaria plantaginifolia* (L.) Hook. | Ant pla | *Coreoposis lanceolata* L. | Cor lan | *Galium circaezans* Michx. | Gal cir |
| *Arabidopsis thaliana* (L.) Heyn. | Ara tha | *Coreopsis palmata* Nutt. | Cor pal | *Galium sp. L.* | Gal spp |
| *Arabis canadensis* L. | Ara can | *Cornus drummondii* Meyer | Cor dru | *Galium virgatum* Nutt. | Gal vir |
| *Arenaria patula* Michx. | Are pat | *Cornus florida* L. | Cor flo | *Geranium carolinianum* L. | Ger car |
| *Arenaria stricta* Michx. | Are str | *Cotinus obovatus* Raf. | Cot obo | *Gerardia flava* L. | Ger fla |
| *Aristida dichotoma* Michx. | Ari dic | *Crataegus sp.* L. | Cra spp | *Gerardia tenuifolia* Vahl | Ger ten |
| *Aristida oligantha* Michx. | Ari oli | *Crotonopsis elliptica* Willd. | Cro ell | *Gnaphalium purpureum* L. | Gna pur |
| *Aristida purpurascens* Poir. | Ari pur | *Crotonopsis linearis* Michx. | Cro lin | *Helianthus divaricatus* L. | Hel div |
| *Asclepias quadrifolia* Jacq. | Asc qua | *Cunila origanoides* (L.) Britt. | Cun ori | *Helianthus hirsutus* Raf. | Hel hir |
| *Asclepias tuberosa* L. | Asc tub | *Cyperus filiculmis* Vahl. | Cyp fil | *Helianthus maximiliani* Schrad. | Hel max |
| *Asclepias verticillata* L. | Asc ver | *Danthonia sericea* Nutt. | Dan ser | *Heliotropium tenellum* (Nutt.) Torr. | Hel ten |
| *Ascryum hypericoides* L. | Asc hyp | *Danthonia spicata* (L.) Beauv. | Dan spi | *Heuchera americana* L. | Heu ame |
| *Asplenium platyneuron* (L.) Oakes. | Asp pla | *Delphinium carolinianum* Walt. | Del car | *Houstonia longifolia* Gaery. | Hou lon |
| *Aster anomalus* Engel. | Ast ano | *Desmodium cuspidatum* (Muhl.) Loud. | Des cus | *Houstonia nigricans* (Lam.) Fern. | Hou nig |
| *Aster drummondii* Lindl. | Ast dru | *Desmodium glutinosum* (Muhl.) Wood | Des glu | *Houstonia purpurea* L. | Hou pur |
| *Aster patens* Ait. | Ast pat | *Desmodium illinoense* Gray | Des ill | *Houstonia tenuifolia* Nuttall. | Hou ten |
| *Aster sp.* L. | Ast spp | *Desmodium laevigatum* (Nutt.) DC. | Des lae | *Hypericum punctatum* L. | Hyp pun |
| *Astragalus canadensis* L. | Ast can | *Desmodium nuttallii* (Schindl.) Schub. | Des nut | *Hypericum spathulatum* (Spach) Steud. | Hyp spa |
| *Baptisia leucophaea* Nutt. | Bap leu | *Desmodium paniculatum* (L.) DC. | Des pan | *Hypericum sphaerocarpum* Michx. | Hyp sph |
| *Botrychium dissectum* Spreng. | Bot dis | *Desmodium pauciflorum* (Nutt.) DC. | Des pau | *Ipomoea sp.* L. | Ipo spp |
| *Bouteloua curtipendula*(Michx.)Torr. | Bou cur | *Desmodium sessilifolium* (Torr.) T. & G. | Des ses | *Juniperus ashei* Buchholz | Jun ash |
| *Bromus purgans* L. | Bro pur | *Desmodium sp.* Desv. | Des spp | *Juniperus virginiana* L. | Jun vir |
| *Bromus racemosus* L. | Bro rac | *Diodia teres* Walt. | Dio ter | *Krigia dandelion (L.)* Nutt. | Kri dan |
| *Bumelia lanuginosa* (Michx.) Pers. | Bum lan | *Dioscorea quaternata* (Walt.) J.F. Gmel. | Dio qua | *Kuhnia eupatorioides* L. | Kuh eup |
| *Carex arctitecta* Mackenz. | Car arc | *Diospyros virginiana* L. | Dio vir | *Lactuca scariola* L. | Lac sca |
| *Carex cephalophora* Muhl. | Car cep | *Echinacea pallida* Nutt. | Ech pal | *Lechea tenuifolia* Michx. | Lec ten |
| *Carex lanuginosa* Michx. | Car lan | *Elymus canadensis* L. | Ely can | *Lepidium virginicum* L. | Lep vir |
| *Carex muhlenbergii* Schk. | Car muh | *Elymus virginicus* L. | Ely vir | *Lespedeza capitata* Michx. | Les cap |
| *Carex retroflexa* Muhl. | Car ret | *Eragrostis spectabilis* (Push) Steud. | Era spe | *Lespedeza hirta* (L.) Hornem. | Les hir |

Appendix 1. Continued

| Genus species | Code | Genus species | Code | Genus species | Code |
|---|---|---|---|---|---|
| *Lespedeza intermedia* (S. Wats.) Britt. | Les int | *Petalostemon candidum* (Willd.) Michx. | Pet can | *Solidago arguta* Ait. | Sol arg |
| *Lespedeza procumbens* Michx. | Les pro | *Petalostemon purpureum* (Vent.) Rydb. | Pet pur | *Solidago nemoralis* Ait. | Sol nem |
| *Lespedeza sp.* Michx. | Les spp | *Phlox pilosa* L. | Phl pil | *Solidago petiolaris* Ait. | Sol pet |
| *Lespedeza virginica* L. | Les vir | *Physalis virginiana* Mill. | Phy vir | *Solidago radula* Nutt. | Sol rad |
| *Liatris aspera* Michx. | Lia asp | *Physostegia virginiana* (L.) Benth. | Phi vir | *Solidago sp.* L. | Sol spp |
| *Liatris cylindracea* Michx. | Lia cyl | *Pinus echinata* Mill. | Pin ech | *Solidago ulmifolia* Muhl. | Sol ulm |
| *Liatris mucronata* DC. | Lia muc | *Plantago virginica* L. | Pla vir | *Sorghastrum nutans* (L.) Nash. | Sor nut |
| *Liatris scabra* (Greene) K. Schum. | Lia sca | *Polygonum convolvulus* L. | Pol con | *Specularia perfoliata* (L.) A. DC. | Spe per |
| *Liatris squarrosa* (L.) Michx. | Lia squ | *Prunus sp.* L. | Pru spp | *Sphenobolis intermedia* Rydb. | Sph int |
| *Linaria canadensis* (L.) Dumort. | Lin can | *Quercus alba* L. | Que alb | *Sphenobolis obtusata* (Michx.) Scribn. | Sph obt |
| *Lithospermum canescens* (Michx.) Lehm. | Lit can | *Quercus falcata* Michx. | Que fal | *Spiranthes gracilis* (Bigel.) Beck | Spi gra |
| *Lithospermum incisum* Lehm. | Lit inc | *Quercus marilandica* Muenchh. | Que mar | *Sporobolus asper* (Michx.) Kunth. | Spo asp |
| *Lobelia inflata* L. | Lob inf | *Quercus prinoides* Willd. | Que pri | *Sporobolus clandestinus* (Biehler) Hitchc. | Spo cla |
| *Lonicera flava* Sims | Lon fla | *Quercus rubra* L. | Que rub | *Strophostyles helvola* (L.) Ell. | Str hel |
| *Matelea decipiens* (Alex.) Woodson | Mat dec | *Quercus stellata* Wang. | Que ste | *Strophostyles leiosperma* (T. & G.) Piper | Str lei |
| *Melica nitens* Nutt. | Mel nit | *Quercus velutina* Lam. | Que vel | *Stylosanthes biflora* (L.) BSP. | Sty bif |
| *Mirabilis albida* (Walt.) Heimerl | Mir alb | *Rhus aromatica* Ait. | Rhu aro | *Symphoricarpos orbiculatus* Moench | Sym orb |
| *Monarda fistulosa* L. | Mon fis | *Rhus copallina* L. | Rhu cop | *Taenidia integerrima* (L.) Drude | Tae int |
| *Muhlenbergia capillaris* (Lam.) Trin. | Muh cap | *Rhus glabra* L. | Rhu gla | *Talinum calycinum* Engelm. | Tal cal |
| *Muhlenbergia sobolifera* (Muhl.) Trin. | Muh sob | *Rhus radicans* L. | Rhu rad | *Talinum parviflorum* Nutt. | Tal par |
| *Myosotis virginica* (L.) BSP. | Myo vir | *Robinia pseudo-acacia* L. | Rob pse | *Tephrosia virginiana* (L.) Pers. | Tep vir |
| *Onosmodium subsetosum* Mackenz. & Bush | Ono sub | *Rosa carolina* (L.) | Ros car | *Tradescantia ohiensis* Raf. | Tra ohi |
| *Opuntia compressa* (Salisb.) Macbr. | Opu com | *Rubus sp.* L. | Rub spp | *Tragia urticifolia* Michx. | Tra urt |
| *Ostrya virginiana* (Mill.) K. Koch | Ost vir | *Rudbeckia hirta* L. | Rud hir | *Tridens flavus* (L.) Hitchc. | Tri fla |
| *Oxalis stricta* L. | Oxa str | *Ruellia humilis* Nutt. | Rue hum | *Ulmus alata* Michx. | Ulm ala |
| *Oxalis violacea* L. | Oxa vio | *Ruellia pedunculata* Torr. | Rue ped | *Ulmus rubra* Muhl. | Ulm rub |
| *Palafoxia callosa* (Nutt.) T. & G. | Pal cal | *Ruellia strepens* L. | Rue str | *Vaccinium stamineum* L. | Vac sta |
| *Panicum boscii* Poir. | Pan bos | *Salvia azurea* Lam. | Sal azu | *Vaccinium vacillans* Torr. | Vac vac |
| *Panicum clandestinum* L. | Pan cla | *Salvia lyrata* L. | Sal lyr | *Verbena bracteata* Lag. & Rodr. | Ver bra |
| *Panicum lanuginosum* Ell. | Pan lan | *Salvia sp.* L. | Sal spp | *Verbena canadensis* (L.) Britt. | Ver can |
| *Panicum latifolium* L. | pan lat | *Sanicula gregaria* Bicknell | San gre | *Verbesina helianthoides* Michx. | Ver hel |
| *Panicum laxiflorum* Lam. | Pan lax | *Sassafras albidum* (Nutt.) Nees | Sas alb | *Verbesina virginica* L. | Ver vir |
| *Panicum linearifolium* Scribn. | Pan lin | *Satureja arkansana* (Nutt.) Briq. | Sat ark | *Vernonia baldwini* Torr. | Ver bal |
| *Panicum oligosanthes* Schult. | Pan oli | *Schrankia uncinata* Wild. | Sch unc | *Viburnum rufidulum* Raf. | Vib ruf |
| *Panicum sp.* L. | Pan spp | *Scleria oligantha* Michx. | Scl oli | *Viola triloba* Schwein. | Vio tri |
| *Panicum virgatum* L. | Pan vir | *Scutellaria ovata* Hill | Scu ova | *Vitis aestivalis* Michx. | Vit aes |
| *Parietaria pensylvanica* Muhl. | Par pen | *Sedum nuttallianum* Raf. | Sed nut | | |
| *Parinychia fastigata* (Raf.) Fern. | Par fas | *Sedum pulchellum* Michx. | Sed pul | | |
| *Parthenocissus quinquefolia* (L.) Planch. | Par qui | *Senecio obovatus* Muhl. | Sen obo | | |
| *Passiflora lutea* L. | Pas lut | *Silene virginica* L. | Sil vir | | |
| *Pellaea atropurpurea* (L.) Link | Pel atr | *Silphium laciniatum* L. | Sil lac | | |
| *Penstemon cobaea* Nutt. | Pen cob | *Sisyrhinchium campestre* Bickn. | Sis cam | | |
| *Penstemon pallidus* Small | Pen pal | *Smilax bona-nox* L. | Smi bon | | |

# VALIDATION OF BEHAVE FIRE BEHAVIOR PREDICTIONS IN OAK SAVANNAS USING FIVE FUEL MODELS

Keith Grabner, John Dwyer, and Bruce Cutter[1]

Abstract: Prescribed fire is a valuable tool in the restoration and management of oak savannas. BEHAVE, a fire behavior prediction system developed by the United States Forest Service, can be a useful tool when managing oak savannas with prescribed fire. BEHAVE predictions of fire rate-of-spread and flame length were validated using four standardized fuel models: Fuel Model 1 (short grass), Fuel Model 2 (timber and grass), Fuel Model 3 (tall grass), and Fuel Model 9 (hardwood litter). Also, a customized oak savanna fuel model (COSFM) was created and validated. Results indicate that standardized fuel model 2 and the COSFM reliably estimate mean rate-of-spread (MROS). The COSFM did not appreciably reduce MROS variation when compared to fuel model 2. Fuel models 1, 3, and 9 did not reliably predict MROS. Neither the standardized fuel models nor the COSFM adequately predicted flame lengths. We concluded that standardized fuel model 2 should be used with BEHAVE when predicting fire rates-of-spread in established oak savannas.

## INTRODUCTION

Historically, Missouri's Ozark forests included an open landscape, containing a thin overstory of trees undergirded with a diverse supply of grasses and wildflowers (Beilmann and Brenner 1951, Ladd 1991, Nigh 1992). It is believed that these savanna-like areas occupied 4.5 to 5.3 million acres throughout the Midwest (Nuzzo 1986) with more than 2 million acres in Missouri (Nelson 1985). There is a renewed interest in reestablishing, restoring, and maintaining oak savannas as part of a diverse forest ecosystem. Land managers are beginning to use large landscape burns for oak savanna restoration.

The use of fire as a restoration and management tool requires managers to develop fire management plans for each savanna. Good fire management plans should include: (1) specific management objectives, (2) control strategies, and 3) methods for assessment of the burn to determine if management objectives were accomplished (Mobely and others 1973).

BEHAVE, a widely accepted fire behavior model developed by the USDA Forest Service, can aid managers in writing fire management plans, since BEHAVE predicts fire behavior such as rate-of-spread, flame length, and fireline intensity (Wilson 1990). The BEHAVE system contains two subsystems: BURN and FUEL (Andrews 1986). BURN contains the programs that predict fire behavior while FUEL is a fuel modeling system. BEHAVE was tested because it can be a valuable planning tool and the model has not been validated in Missouri, although, management experience suggests that BEHAVE does adequately predict fire behavior.

The inputs for BEHAVE are fuel, site, and weather information. The fuel information for BEHAVE comes from a fuel model. This model is a description of the fuel in an area, generally in tons per acre. BEHAVE can use either a standardized or customized fuel model for fuel information (Andrews 1986). There are 13 standardized fuel models which describe average fuel conditions in common fuel types (Anderson 1982). Customized fuel models are developed for fuel conditions that are not adequately described by the standardized fuel models (Burgan and Rothermel 1984). At first glance, fuels in Missouri oak savannas appear to be described by standardized fuel models 1 (short grass), 2 (timber and grass), 3 (tall grass), and 9 (hardwood litter). These standardized fuel models were used to test BEHAVE predictions in established oak savannas.

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[1]Grabner is a former graduate assistant, while both Dwyer and Cutter are Associate Professors of Forestry, School of Natural Resources, 1-30 Agriculture Building, University of Missouri-Columbia, Columbia, MO 65211.

The purpose of this project was to validate fuel models 1 (short grass), 2 (timber and grass), 3 (tall grass), and 9 (hardwood litter) for calculating prescribed fire behavior in established oak savannas in Missouri and develop and validate a customized oak savanna fuel model. There were two reasons for creating a customized oak savanna fuel model: (1) To describe the established savanna fuel conditions, and (2) to reduce the variability between predicted and observed fire behavior.

Fire behavior predictions were validated in established savannas because these communities are managed with prescribed fire and there is an increasing interest in restoring this community in Missouri. Established sites were chosen over restoration sites because it was felt that established savannas would maintain a more constant fuel makeup than a site undergoing restoration. For the scope of this project an established savanna is considered any area that has an overstory crown closure between 10 and 80%, a shrub layer is either present or absent, the herbaceous layer contains both prairie and forest species, and the area is managed with fire (Nuzzo 1986).

## METHODS

### Site Selection

In the spring of 1994, lists of potential savanna study sites were acquired from the Missouri Department of Conservation, Missouri Department of Natural Resources, the United States Forest Service, and The Nature Conservancy. Savanna sites were visited in the summer of 1994 and compared to our definition of an established savanna. Eight established savanna sites were selected (Figure 1).

### Data Collection

Required data consisted of BEHAVE input and output data which was broken into pre-burn and burn treatment data. Slope, aspect, and fuel loading data were collected pre-burn, and weather, fuel moisture, and fire behavior data were collected during each prescribed fire.

Pre-burn data were collected from a series of nested transects. Sampling points were randomly located on each study site. A sampling point consisted of a 132 ft fire behavior transect, oriented perpendicular to the slope. Slope and aspect data were collected along this fire behavior transect. Located at the origin and endpoint of each fire behavior transect were four down dead woody (DDW) fuel transects. The DDW fuel transects were orientated at $0^{o}$, $90^{o}$, $180^{o}$, and $270^{o}$ from the origin and endpoints of the fire behavior transect and were 50-foot long.

Fuel loading was estimated using a technique described by Brown (1984) and Brown and others (1982). Down dead woody fuels that intersected the fuel transect were sampled by timelag class: 1-hr fuels (0 - 0.25 in. diameter), 10-hr fuels (0.25 - 1.0 in. diameter), and 100-hr fuels (1.0 - 3.0 in. diameter). Timelag class is related to the rate which the fuel gain or loses moisture. The woody fuel data were converted to fuel weights using calculations described by Brown (1984).

Herbaceous fuels were sampled using a 1 $ft^2$ clipping frame located at the end of each fuel transect. Herbaceous fuels were clipped and taken to a laboratory where the fuels were dried and the herbaceous fuel loadings were estimated. Litter weights were estimated from litter depths following procedures described by Burgan and Rothermel (1984).

Burn treatment data were collected during the prescribed burns from the edge of each prescribed fire and the fire behavior transects. Weather data included relative humidity, wind speed, and wind direction. Weather data were collected using a belt weather kit along the fire line at each study site.

Fuel moisture data was estimated from fuel samples collected near the prescribed fire site. 1-hr and 10-hr fuel samples were collected along the fire line during each prescribed fire. These samples were collected hourly, placed in air tight containers, and brought back to the laboratory. Fuel moisture samples were weighed, dried, and fuel moisture content was calculated.

Figure 1. Location of all study sites.

Fire behavior data was collected remotely from the fire behavior transects during each prescribed fire. Fire behavior data collected included fire rate-of-spread and flame height. Rate-of-spread data was collected using a rate-of-spread clock (ROSC) system described by Blank and Simard (1983). The rate-of-spread clocks were modified digital watches, wired to a circuit board, and designed to be activated by a flaming front. Clocks were located at the origins and endpoints of each fire behavior transect, and an additional clock was located at $90^0$ and 50 ft from the origin of each transect. Using three clocks to measure rate-of-spread aloud triangulation of the fire's direction of spread and rate-of-spread (Simard and others 1984).

Flame length data was collected using char height as a surrogate. Substituting char height for flame length can be a source of error due to the inherent differences between the two measures. Flame length is the distance from the active flaming zone to the approximate tip of the flaming front. Char height is a measure of flame height, the distance from the ground to the tip of the flaming front. Under burning conditions were there is no wind and the ground is level the flame height can be considered equal to the flame length. If there is wind or a slope tilting the flaming front then the flame length will be greater than the flame height. This difference between flame length and flame height is a source of error when using flame height as a surrogate for flame length. Char height as a measure of flame height can under estimate flame length up to 50% (Cain 1984). The error between char height and flame length was considered acceptable due to logistical and budgetary constraints. A flame length monitoring technique was needed that could monitor multiple fire behavior transects at one time, have a quick setup time, and fit within the budget of this project. Other flame length monitoring schemes were evaluated but char height best met the needs of this project.

Char height was measured on trees surrounding the origin and endpoint of each fire behavior transect. The char height data were used as an estimate of flame length and the potential 50% underestimation was acknowledged.

A customized oak savanna fuel model (COSFM) was created using the mean fuel data from each savanna site. Fuel model construction is not an exact science and variables within a fuel model were adjusted to calibrate the model (Burgan and Rothermel 1984).

Slope, weather, and fuel moisture data were used as inputs for BEHAVE predictions (Table 1). Fuel models 1, 2, 3, 9, and the COSFM were used with BEHAVE to predict fire behavior. All fire behavior predictions were influenced by the assumptions of the BEHAVE model. The assumptions of BEHAVE are that BEHAVE predicts fire behavior in the active flaming front, the fire is advancing in a steady state independent of the ignition source, the fire is spreading through surface fuels, and fuel, fuel moisture, wind, and slope are constant during the prediction period (Rothermel 1983, Andrews 1986). Fire behavior predictions are subjected to increased error when one or more of the assumptions are violated.

Fire behavior predictions were then compared to the actual fire behavior (Table 2) and the models were validated. Model validation is evaluating a model's predictions to see if they are acceptable, validation is not determining if the model is correct or not (Burk 1986, Vanclay 1995). BEHAVE predictions were validated using simple linear regression analysis (Brown 1972, Sneeuwjagt and Frandsen 1977, Andrews 1980, Brown 1982, Norum 1982, Bushey 1985, Hunt and Cook 1987, van Wilgen and Wills 1988, McAlpine and Xanthopoulos 1989). Observed fire behavior was compared to predicted fire behavior and the slope and intercept of the regression equation was analyzed. Given the variation within the data if an equation had a slope equal to one ($H_o$: $\beta_1 = 1$) and an intercept equal to zero ($H_o$: $\beta_0 = 0$), the model predictions could not be distinguished from observed fire behavior, (Smith and Rose 1995). $R^2$s were compared for each equation to identify how well the line fit the data (Neter and others 1989). Regression analysis was completed on mean fire behavior by site (Table 2) due to the large variability expected between individual observed and predicted fire behavior (Andrews 1985, Bushey 1985).

## RESULTS AND DISCUSSION

BEHAVE predictions using fuel models 1, 3, and 9 did not reliably predict prescribed fire MROS, but fuel model 2 did reliably predict fire MROS in established oak savannas (Table 3, Figure 2). Fuel model 2 passed the

Table 1. Input data for fire behavior predictions

| Plot | Slope | Aspect | Relative humidity | Wind | | Fuel Moisture | | |
|---|---|---|---|---|---|---|---|---|
| | | | | Speed | Direction | 1-Hour | 10-Hour | 100-Hour |
| | percent | | percent | mph | | ----------percent---------- | | |
| Knob Noster State Park | | | | | | | | |
| A | 3 | 350° | 42 | 4.0 | 175° | 8.0 | 9.0 | 9.0 |
| B | 7 | 333° | 42 | 6.0 | 183° | 8.0 | 9.0 | 9.0 |
| C | 9 | 45° | 43 | 4.0 | 180° | 8.0 | 9.0 | 9.0 |
| D | 2 | 140° | 43 | 4.0 | 180° | 8.0 | 9.0 | 9.0 |
| E | 2 | 353° | 42 | 5.0 | 180° | 8.0 | 9.0 | 9.0 |
| Ha Ha Tonka State Park | | | | | | | | |
| A | 15 | 291° | 33 | 3.0 | 240° | 8.5 | 20 | 20 |
| B | 10 | 343° | 30 | 2.0 | 250° | 8.5 | 20 | 20 |
| C | 21 | 337° | 32 | 3.0 | 220° | 8.5 | 20 | 20 |
| D | 24 | 17° | 32 | 2.5 | 250° | 8.5 | 20 | 20 |
| E | 15 | 58° | 32 | 2.5 | 250° | 8.5 | 20 | 20 |
| Meremac State Park | | | | | | | | |
| A | 4 | 113° | 40 | 3.0 | 295° | 11 | 14 | 11 |
| B | 20 | 182° | 41 | 3.0 | 303° | 11 | 14 | 11 |
| C | 21 | 194° | 39 | 3.0 | 295° | 11 | 14 | 11 |
| D | 21 | 181° | 40 | 3.5 | 336° | 11 | 14 | 11 |
| G | 12 | 215° | 41 | 0.0 | | 11 | 14 | 11 |
| Taum Sauk State Park | | | | | | | | |
| A | 5 | 7° | 54 | 6.0 | 200° | 8 | 9 | 10 |
| B | 5 | 50° | 54 | 6.0 | 200° | 8 | 9 | 10 |
| D | 5 | 350° | 54 | 6.0 | 200° | 8 | 9 | 10 |
| P-Highway | | | | | | | | |
| B | 13 | 219° | 28 | 2.0 | 35° | 8 | 20 | 20 |
| E | 8 | 237° | 29 | 6.0 | 360° | 8 | 20 | 20 |
| University State Forest | | | | | | | | |
| A | 1 | 290° | 45 | 1.0 | 340° | 8 | 12 | 12 |
| C | 1 | 360° | 50 | 0.0 | | 8 | 12 | 12 |
| Peck Ranch Conservation Area | | | | | | | | |
| B | 5 | 306° | 35 | 8.0 | 210° | 4 | 7 | 13 |
| C | 12 | 105° | 28 | 1.0 | 220° | 4 | 7 | 13 |
| D | 15 | 270° | 29 | 7.0 | 225° | 4 | 7 | 13 |
| E | 12 | 104° | 28 | 3.0 | 220° | 4 | 7 | 13 |
| F | 13 | 310° | 28 | 3.0 | 220° | 4 | 7 | 13 |
| G | 16 | 250° | 28 | 3.0 | 225° | 4 | 7 | 13 |
| H | 14 | 232° | 28 | 3.0 | 230° | 4 | 7 | 13 |
| I | 9 | 230° | 28 | 3.0 | 220° | 4 | 7 | 13 |
| J | 8 | 250° | 28 | 3.0 | 220° | 4 | 7 | 13 |

Table 2. Predicted and observed fire behavior for all sites.

| Plot | Predicted rate-of-spread | | | | observed | Predicted flame height | | | | observed |
|---|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 9 | | 1 | 2 | 3 | 9 | |
| | chains/hour | | | | | feet | | | | |
| Knob Noster State Park | | | | | | | | | | |
| A | 18.0 | 7.0 | 26.0 | 2.0 | 7.0 | 5.2 | 7.8 | 13.6 | 3.1 | 3.0 |
| B | 43.0 | 18.0 | 64.0 | 4.0 | 13.0 | 2.7 | 4.3 | 8.4 | 1.8 | 2.0 |
| C | 22.0 | 9.0 | 29.0 | 2.0 | 30.0 | 2.3 | 3.7 | 6.9 | 1.5 | 5.0 |
| D | 38.0 | 16.0 | 56.0 | 4.0 | 30.0 | 2.2 | 3.6 | 7.0 | 1.5 | 4.0 |
| E | 76.0 | 36.0 | 96.0 | 7.0 | 40.0 | 4.0 | 6.3 | 11.8 | 2.6 | 5.0 |
| Mean | 39.4 | 17.2 | 54.2 | 3.8 | 24.0 | 3.3 | 5.1 | 9.5 | 2.1 | 4.0 |
| Ha Ha Tonka State Park | | | | | | | | | | |
| A | 13.0 | 6.0 | 21.0 | 1.0 | 8.0 | 0.7 | 1.4 | 3.1 | 0.6 | 1.0 |
| B | 30.0 | 13.0 | 53.0 | 4.0 | 13.0 | 0.8 | 1.7 | 3.8 | 0.8 | 1.0 |
| C | 28.0 | 13.0 | 52.0 | 3.0 | 30.0 | 1.2 | 2.5 | 5.1 | 1.1 | 1.0 |
| D | 26.0 | 12.0 | 50.0 | 3.0 | 9.0 | 1.4 | 2.8 | 5.8 | 1.2 | 0.5 |
| E | 30.0 | 11.0 | 33.0 | 3.0 | 4.0 | 1.9 | 4.0 | 8.4 | 1.7 | 1.0 |
| Mean | 25.4 | 11.0 | 41.8 | 2.8 | 12.8 | 1.2 | 2.5 | 5.2 | 1.1 | 1.0 |
| Meremac State Park | | | | | | | | | | |
| A | 8.0 | 5.0 | 15.0 | 1.0 | 10.0 | 1.5 | 2.6 | 5.0 | 1.0 | 0.5 |
| B | 10.0 | 5.0 | 9.0 | 1.0 | 22.0 | 1.5 | 2.7 | 4.1 | 1.1 | 1.0 |
| C | 11.0 | 5.0 | 17.0 | 1.0 | 4.0 | 1.7 | 2.4 | 5.3 | 1.1 | 1.0 |
| D | 11.0 | 6.0 | 17.0 | 1.0 | 4.0 | 1.7 | 2.9 | 5.3 | 1.1 | 0.5 |
| G | 6.0 | 4.0 | 6.0 | 1.0 | 7.0 | 1.2 | 2.2 | 3.3 | 1.0 | 1.0 |
| Mean | 9.2 | 5.0 | 12.8 | 1.4 | 11.8 | 1.5 | 2.6 | 4.6 | 1.1 | 1.0 |
| Taum Sauk Mountain State Park | | | | | | | | | | |
| A | 18.0 | 7.0 | 20.0 | 2.0 | 5.0 | 2.1 | 2.9 | 5.7 | 1.3 | 4.0 |
| B | 30.0 | 11.0 | 33.0 | 3.0 | 15.0 | 2.6 | 3.6 | 7.2 | 1.6 | 5.0 |
| D | 11.0 | 4.0 | 12.0 | 1.0 | 4.0 | 1.7 | 2.3 | 4.6 | 1.0 | 6.0 |
| Mean | 19.7 | 7.3 | 21.7 | 2.0 | 8.0 | 2.1 | 2.9 | 5.8 | 1.3 | 5.0 |
| P-Highway | | | | | | | | | | |
| B | 6.0 | 3.0 | 6.0 | 1.0 | 6.0 | 1.3 | 2.1 | 3.4 | 1.0 | 3.0 |
| E | 20.0 | 9.0 | 22.0 | 2.0 | 7.0 | 2.2 | 3.3 | 5.9 | 1.3 | 5.0 |
| Mean | 13.0 | 6.0 | 14.0 | 1.5 | 6.5 | 1.8 | 2.7 | 4.7 | 1.2 | 4.0 |
| University State Forest | | | | | | | | | | |
| A | 0.0 | 2.0 | 13.0 | 1.0 | 1.0 | 0.0 | 1.6 | 4.6 | 1.0 | 0.5 |
| C | 0.0 | 1.0 | 3.0 | 0.0 | 1.0 | 0.0 | 0.8 | 2.4 | 0.5 | 1.0 |
| Mean | 0.0 | 1.5 | 8.0 | 0.5 | 1.0 | 0.0 | 1.2 | 3.5 | 0.8 | 1.0 |
| Peck Ranch Conservation Area | | | | | | | | | | |
| B | 20.0 | 7.0 | 19.0 | 2.0 | 5.0 | 2.3 | 3.0 | 6.1 | 1.4 | 2.0 |
| C | 10.0 | 5.0 | 20.0 | 2.0 | 4.0 | 1.7 | 2.5 | 6.3 | 1.4 | 4.0 |
| D | 14.0 | 4.0 | 13.0 | 1.0 | 28.0 | 2.3 | 2.7 | 5.8 | 1.3 | 2.0 |
| E | 43.0 | 18.0 | 78.0 | 5.0 | 13.0 | 3.3 | 4.7 | 11.7 | 2.4 | 5.0 |
| F | 11.0 | 5.0 | 20.0 | 1.0 | 5.0 | 1.8 | 2.6 | 6.2 | 1.3 | 2.0 |
| G | 9.0 | 4.0 | 18.0 | 1.0 | 1.0 | 1.6 | 2.4 | 6.0 | 1.2 | 2.0 |
| H | 4.0 | 2.0 | 8.0 | 1.0 | 2.0 | 1.1 | 1.6 | 4.0 | 0.8 | 2.0 |
| I | 20.0 | 8.0 | 37.0 | 2.0 | 2.0 | 2.3 | 3.4 | 8.3 | 1.7 | 3.0 |
| J | 26.0 | 11.0 | 48.0 | 3.0 | 9.0 | 2.6 | 3.8 | 9.4 | 2.0 | 3.0 |
| Mean | 17.4 | 7.1 | 29.0 | 2.0 | 7.7 | 2.9 | 2.9 | 7.1 | 1.5 | 2.8 |

slope and intercept tests ($\alpha = 0.05$) and inspection of the regression line through the data reveals a good fit between the data and the regression for MROS predictions (Figure 2). MROS predictions by site were less than +/- four chains per hour from the mean observed rate-of-spread data by site. All the standardized fuel models were unreliable for predicting flame heights (Table 3, Figure 3). Fuel models 1, 2, and 3 tended to overpredict mean flame height and fuel model 9 tended to underpredict mean flame heights.

Four standardized fuel models were validated for use in established oak savannas, since BEHAVE users need to be flexible when deciding what fuel model will work best in a fuel type (Norum 1982). Although, a fuel model appears to characterize fuel conditions in an area does not mean that it will produce acceptable fire behavior predictions (Andrews 1986). Four standardized fuel models which appear to at least partially describe savanna fuels were chosen to predict prescribed fire behavior. Fuel models 1 (short grass), 3 (tall grass), and 9 (hardwood litter) did not adequately predict prescribed MROS or mean flame heights. This indicates that these fuel models do not fit established oak savanna fuel conditions. Fire behavior predictions that have large deviations from observed fire behavior indicate that the fuel model used does not adequately represent the fuels in a fuel type (McAlpine and Xanthopoulos 1989).

Fuel model 2 (timber and grass) provided reasonable predictions of MROS but not mean flame height. The mean predictions for fuel model 2 did tend to under predict rate-of-spread but this is expected when working in heterogeneous fuels (Catchpole and others 1993). Comparison of rate-of-spread predictions from fuel model 2 to similar studies shows the results to be consistent with other BEHAVE validation work (Table 4).

A customized oak savanna fuel model (COSFM) was created using the mean fuel loading for all study sites. The fuel weights for the customized fuel model were compared to all the savanna sites to test how well the model represented the sites (Table 5). The fuel weights for P-Highway Savanna in the 10- and 100-hr fuels were statistically different ($\alpha=0.05$) from the customized fuel model. This site underwent a firewood cutting. Tops from harvested trees increased the fuel loads in the 10- and 100-hr fuels. Taum Sauk Savanna was statistically different ($\alpha=0.05$) from the customized fuel model in the 1- and 10-hr fuel classes. The area used within Taum Sauk Mountain was at one time a pasture with some widely spaced trees. The savanna at Meremac State Park was statistically different from the customized fuel model in the 100-hr fuel class. This was because this site has the lowest number of trees per acre and basal area of all the study sites. There were fewer trees contributing larger branch material to the 100-hr fuel size class.

Once the fuel weights for the different fuel size classes were calculated, a customized fuel model was developed (Table 6). MROS predictions using the COSFM were reasonable (Table 3, Figure 2). The COSFM passed the slope and intercept test ($\alpha=0.05$). The error for the MROS predictions for all sites was less than +/- four chains per hour. The mean flame height prediction of the COSFM was not reliable and tended to overpredict mean flame height (Table 3, Figure 3).

A customized oak savanna fuel model (COSFM) was created in an attempt to reduce some of the variability between fire behavior predictions and observations. The COSFM did slightly reduce the variability, making mean fire behavior predictions closer to observed fire behavior. The COSFM reliability was comparable to past validation work (Table 4). However, considering the apparent reliability of fuel model 2 and the amount of time required to collect and analyze fuel data, it was concluded that a COSFM was not needed for the selected established oak savanna sites.

Fuel model 2 was the best standardized fuel model for the established oak savanna sites tested. With good input data and appropriate interpretation of the output, BEHAVE can be a useful management tool (Andrews 1980). Because BEHAVE is computer operated, it is easy to accept its predictions as accurate (Andrews and Latham 1984), but one should keep in mind that BEHAVE is not a substitute for experience (Rothermel 1983). If used carefully BEHAVE can produce acceptable fire behavior predictions and can be a useful tool when managing Missouri's oak savannas with prescribed fire.

Table 3. Simple linear regression ANOVA table for mean fire behavior predictions

| Fuel model | Rate-of-spread | | | | Flame height | | | |
|---|---|---|---|---|---|---|---|---|
| | Source of variation | SS | df. | MS | Source of variation | SS | df. | MS |
| 1 (Short Grass) | Regression | 185.88 | 1 | 185.88 | Regression | 5.45 | 1 | 5.45 |
| | Error | 43.05 | 5 | 8.61 | Error | 11.70 | 6 | 2.34 |
| | Total | 228.93 | 6 | | Total | 17.15 | 7 | |
| | $R^2 = 0.81$ | $Y = 1.855 + 0.350(X)$ | | | $R^2 = 0.32$ | $Y = 0.587 + 0.916(X)$* | | |
| 2 (Timber and Grass) | Regression | 193.12 | 1 | 193.12 | Regression | 4.87 | 1 | 4.87 |
| | Error | 35.81 | 5 | 7.16 | Error | 12.29 | 6 | 2.46 |
| | Total | 228.93 | 6 | | Total | 17.15 | 7 | |
| | $R^2 = 0.84$ | $Y = 0.975 + 0.928(X)$ | | | $R^2 = 0.28$ | $Y = -0.096 + 0.828(X)$* | | |
| 3 (Tall Grass) | Regression | 204.27 | 1 | 204.27 | Regression | 2.29 | 1 | 2.29 |
| | Error | 24.66 | 5 | 4.93 | Error | 14.87 | 6 | 2.97 |
| | Total | 228.93 | 6 | | Total | 17.15 | 7 | |
| | $R^2 = 0.89$ | $Y = 0.104 + 0.293(X)$ | | | $R^2 = 0.13$ | $Y = 0.405 + 0.304(X)$* | | |
| 9 (Hardwood Litter) | Regression | 179.33 | 1 | 179.33 | Regression | 4.17 | 1 | 4.17 |
| | Error | 49.59 | 5 | 9.92 | Error | 12.99 | 6 | 2.60 |
| | Total | 228.93 | 6 | | Total | 17.15 | 7 | |
| | $R^2 = 0.78$ | $Y = 0.592 + 3.876(X)$ | | | $R^2 = 0.24$ | $Y = -0.224 + 1.850(X)$* | | |
| Customized Fuel Model | Regression | 194.02 | 1 | 194.02 | Regression | 3.82 | 1 | 3.82 |
| | Error | 34.91 | 5 | 34.91 | Error | 13.34 | 6 | 2.67 |
| | Total | 228.93 | 6 | | Total | 17.15 | 7 | |
| | $R^2 = 0.85$ | $Y = 0.651 + 1.064(X)$ | | | $R^2 = 0.22$ | $Y= -0.068 + 0.773(X)$* | | |

Y = Observed fire behavior
X = Predicted fire behavior
* = No relationship between Y and X ($\alpha$=0.05)

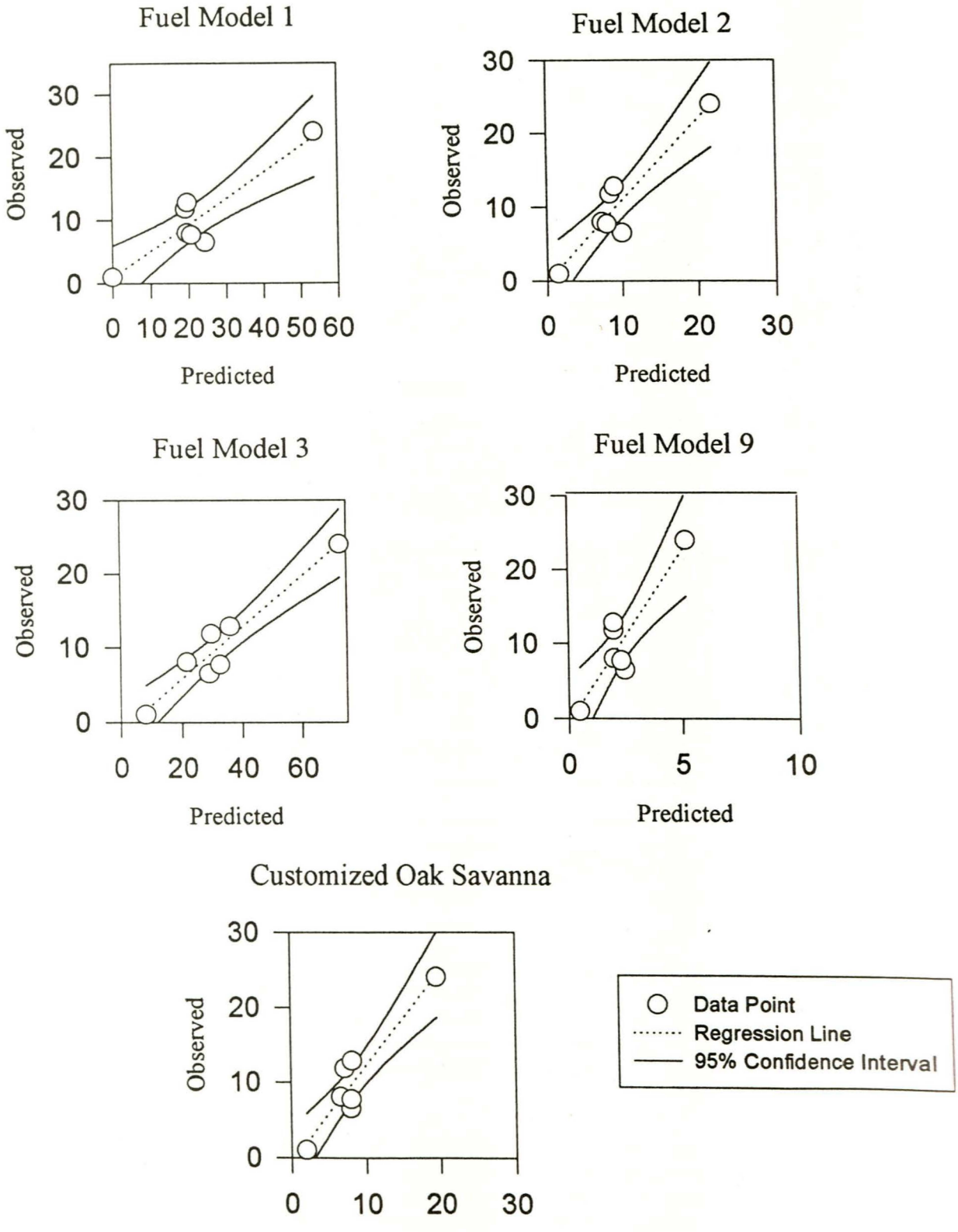


Figure 2. Mean predicted vs. mean observed rates-of-spread for each fuel model.

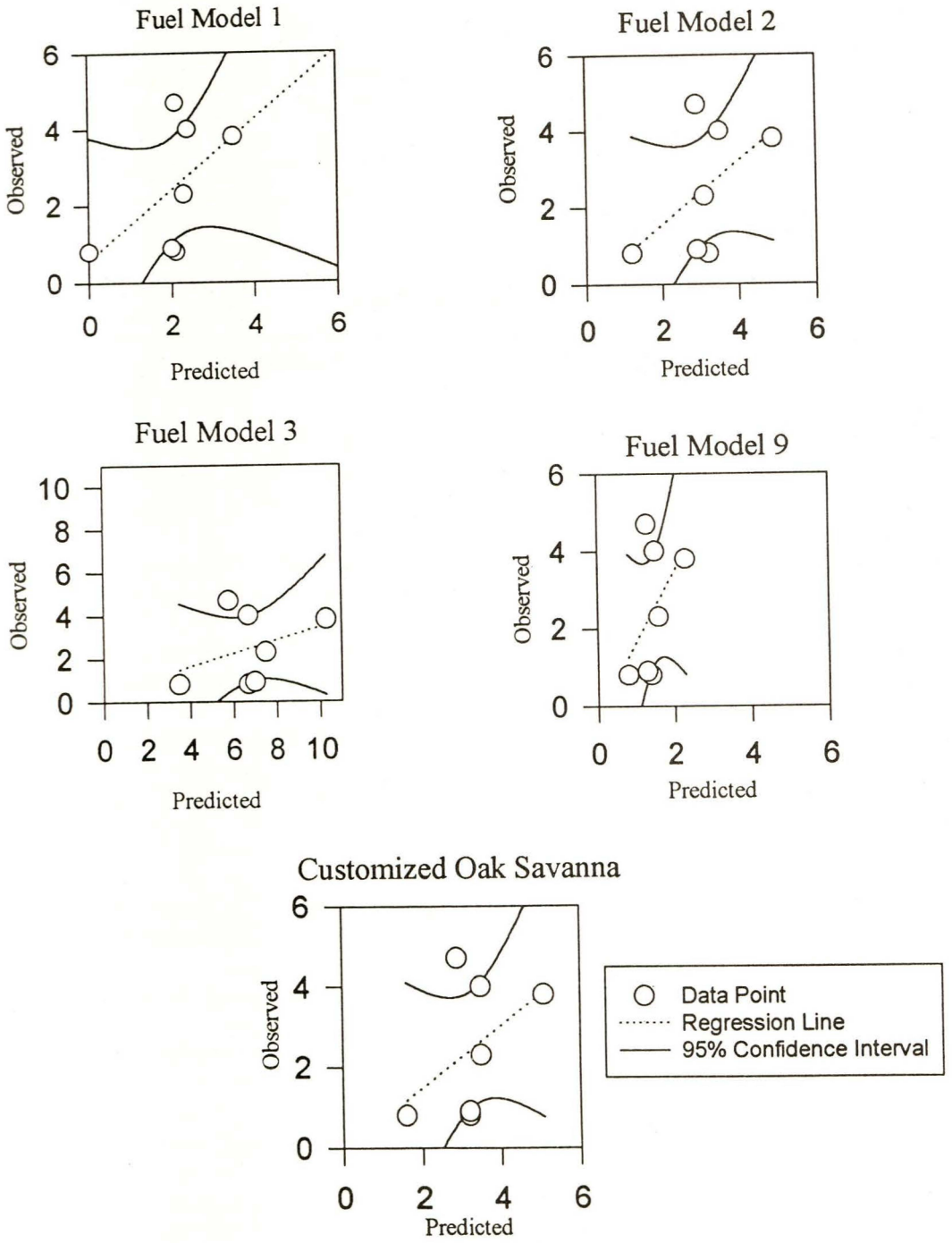


Figure 3. Mean predicted vs. mean observed flame length for each fuel model.

Table 4. Comparison of fuel model 2 validation in oak savannas to other BEHAVE validation work

| Fuel type | $R^2$ | Citation |
|---|---|---|
| Oak Savanna: | | |
| Predictions using COSFM | 0.85 | Grabner 1996 |
| Predictions using Fuel Model 2 | 0.84 | Grabner 1996 |
| Grass | 0.92 | Sneeuwjagt and Frandsen 1977 |
| Western Slash | 0.80 | Brown 1972 |
| Southern Rough | 0.89 | Hough and Albini 1978 |
| Sage Brush | 0.60 | Bushey 1985 |
| Black Spruce | 0.94 | Norum 1982 |
| Ponderosa Pine | 0.78 | McAlpine and Xanthopoulos 1989 |
| Synthesis of Validation Research up to 1980 | 0.89 | Andrews 1980 |
| South African Savanna | 0.82 | van Wilgen and Wills 1988 |

Table 5. Comparison of fuel weights by site to customized fuel model

| Site | Litter | Herbaceous | 1-hour | 10-hour | 100-hour |
|---|---|---|---|---|---|
| | ----------tons/acre---------- | | | | |
| Knob Noster S.P. | 3.02 | 0.82a | 0.044 | 0.297 | 1.672 |
| Ha Ha Tonka S.P. | 3.12 | 0.33 | 0.100 | 0.364 | 0.543 |
| Meremac S.P. | 2.50 | 0.48 | 0.042 | 0.220 | 0.253a |
| Taum Sauk S.P. | 2.90 | 0.51 | 0.012a | 0.159a | 0.317 |
| Bennett Spring | 2.56 | 0.35 | 0.053 | 0.573 | 0.709 |
| P-Highway | 2.72 | 0.08a | 0.100 | 0.675a | 2.896a |
| University State Forest | 2.18 | 0.09 | 0.071 | 0.425 | 1.394 |
| Peck Ranch Conservation Area | 2.76 | 0.78 | 0.053 | 0.477 | 2.111 |
| Custom Oak Savanna Fuel Model | 2.72 | 0.43 | 0.058 | 0.398 | 1.236 |

a = indicates values that are significantly different from the customized fuel model ($\alpha$=0.05).

Table 6. Structure of customized oak savanna fuel model.

| Fuel load | | Surface area to volume ratio | | Other | |
|---|---|---|---|---|---|
| ----------tons/acre---------- | | | | | |
| 1 Hr | 3.21 | 1 Hr | 2800 | Depth (ft) | 0.63 |
| 10 Hr | 0.40 | Live Herb. | 0 | Heat Content (BTU/lb.) | 8000 |
| 100 Hr | 1.24 | Live Woody | 0 | Extinction Moisture (%) | 30 |
| Live Herb | 0.00 | | | | |
| Live Woody | 0.00 | | | | |

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# SPATIAL CONFIGURATION AND DISTRIBUTION OF FOREST PATCHES IN CHAMPAIGN COUNTY, ILLINOIS: 1940 TO 1993

J. Danilo Chinea[1]

Abstract: Spatial configuration and distribution of landscape elements have implications for the dynamics of forest ecosystems, and, therefore, for the management of these resources. The forest cover of Champaign County, in east-central Illinois, was mapped from 1940 and 1993 aerial photography and entered in a geographical information system database. In 1940, 208 forest patches amounted to 1,998 ha (or 0.8% of the county area). These patches had a mean area of 10 ha, a mean perimeter of 2,016 m, a mean perimeter to area ratio of 330, a fractal dimension of 1.47, and a mean nearest neighbor distance of 391 m. In 1993, 116 forest patches covered 3,370 ha (or 1.3% of the county area). These patches had a mean area of 29 ha, a mean perimeter of 4,272 m, a mean perimeter to area ratio of 247, a fractal dimension of 1.56, and a mean nearest neighbor distance of 558 m. However, only 179 ha and 480 ha can be considered interior forest (i.e., forest area within 100 m from the edges) in 1940 and 1993, respectively. In both years, almost all forest patches were located along the major rivers and streams of the county. During this 53 year period, the forest patches of this county have increased in size and shape complexity, and at the same time have become less fragmented. These changes resulted mostly from natural reforestation in portions of land adjacent to older forest patches, even though the number of forest patches was reduced in this period. Although these changes should result in improvements in habitat characteristics for the forest biota, this is still a very fragmented landscape and management alternatives should consider the spatial characteristics of these forest patches.

Alfisols accounted for the largest proportions in forest cover among the soil orders present in the county in both years (7% and 10%, respectively). However, the soil series with the largest proportions in forest cover at any of these two dates were two Mollisol series with severe flooding problems: the Ross and Colo series. These results suggest that farming restrictions have had an important impact in forest cover changes during this period.

[1] Assistant Professor, Department of Natural Resources and Environmental Sciences, University of Illinois, 1011 Plant Science Lab., 1201 S. Dorner Dr., Urbana, IL 61801.

# THE MIDWEST FLOOD OF 1993: DID TREES PROTECT LEVEES ALONG THE MISSOURI RIVER?

John P. Dwyer, Douglas Wallace, and David R. Larsen[1]

Abstract: Following the Midwest flood of 1993, a study was initiated along a 39-mile segment of the Missouri River to determine if there was an association between woody corridors and levee stability. A systematic sample of levee failures revealed that primary levees which did not fail had a significantly wider woody corridor than failed levees. Analysis of the total inventory of failed levees revealed that as the width of the woody corridor decreased, the length of the levee failure increased. Number of levee failures and their severity of damage could be reduced if woody corridors were at least 300 feet wide.

[1] Associate Professor, The School of Natural Resources, University of Missouri, 1-30 Agriculture Building, Columbia, MO 65211; Forester, USDA Natural Resources Conservation Service, State Office Parkade Center, Columbia, MO 65203; Assistant Professor, The School of Natural Resources, University of Missouri, 1-30 Agriculture Bldg., Columbia, MO 65211, respectively.

# THREE-YEAR BOLE RESPONSE OF WHITE OAK (*QUERCUS ALBA* L.) CROP TREES TO FERTILIZER AND CROWN RELEASE ON A TENNESSEE UPLAND SITE

G. Richard Schaertl[1], Allan E. Houston[2], Edward R. Buckner[3], and James S. Meadows[4]

Abstract: Bole diameter responses to fertilization, crown release, and fertilization x release treatments with untreated controls and treatment-by-year interactions were studied in pole-sized (approx. 43 years old) white oak (*Quercus alba* L.) crop trees. In the main study, fertilizer was applied by broadcast to plots at a rate of 150 lbs N and 35 lbs $P_2O_5$ per acre, and crown release removed competing trees around crop trees—dominate or codominate trees selected for favorable timber characteristics—in randomly selected plots during spring 1993. Another fertilizer application was applied during spring 1995 at the same rate and method as the first application. Bole diameter, measured at 4.5 ft above ground line (dbh); scaling diameter inside-bark (sdib) and radial increment, both variables measured at 17.3 ft above ground; form class; and stem taper were examined. Dbh, sdib, increment, and form class were measured for 4 years beginning with 1992 pretreatment data; stem taper was measured for 1995 only. Stem taper and form class did not differ significantly among treatments (Table 1). Dbh differences among treatments were not significant; however, pooled mean dbh increased significantly each year from 9.44 inches in 1992 to 10.29 inches in 1995 (2.9% compound growth rate). Sdib differences among treatments also were not significant, but pooled mean sdib increased from 7.28 inches in 1992 to 7.97 inches in 1995 (3.1% compound growth rate). At the end of the 1993 growing season, radial increments for release (0.12 in.) and fertilization x release (0.12 in.) treatments were significantly greater than the control (0.09 in.). The fertilization x release treatment continued to significantly increase radial increment (0.14 in.) more than fertilization and control treatments (0.11 in., both treatments) in 1994. At the end of the 1995 growing season, radial increment for fertilization x release (0.16 in.) was significantly greater than release, fertilization, and control (0.12 in., 0.11 in., 0.10 in., respectively). Fertilization x release treatment provided greater cambial increment for the first 3 years on mid-rotation white oak crop trees. Should trends continue, the fertilization x release treatment will improve volume in the first log and may improve log form more than release, fertilization, and the control treatments.

[1] Assistant Professor and Forest Management Specialist, Department of Forestry, Wildlife & Fisheries, The University of Tennessee, Jackson, TN 38301-3201.

[2] Research Assistant Professor, Ames Plantation Experiment Station, The University of Tennessee, Grand Junction, TN 38039-0389.

[3] Distinguished Professor, Forest Management, Department of Forestry, Wildlife & Fisheries, The University of Tennessee, Knoxville, TN 37901-1071.

[4] Principal Silviculturist, USDA Forest Service, Southern Research Station, Southern Hardwoods Laboratory, Stoneville, MS 38776.

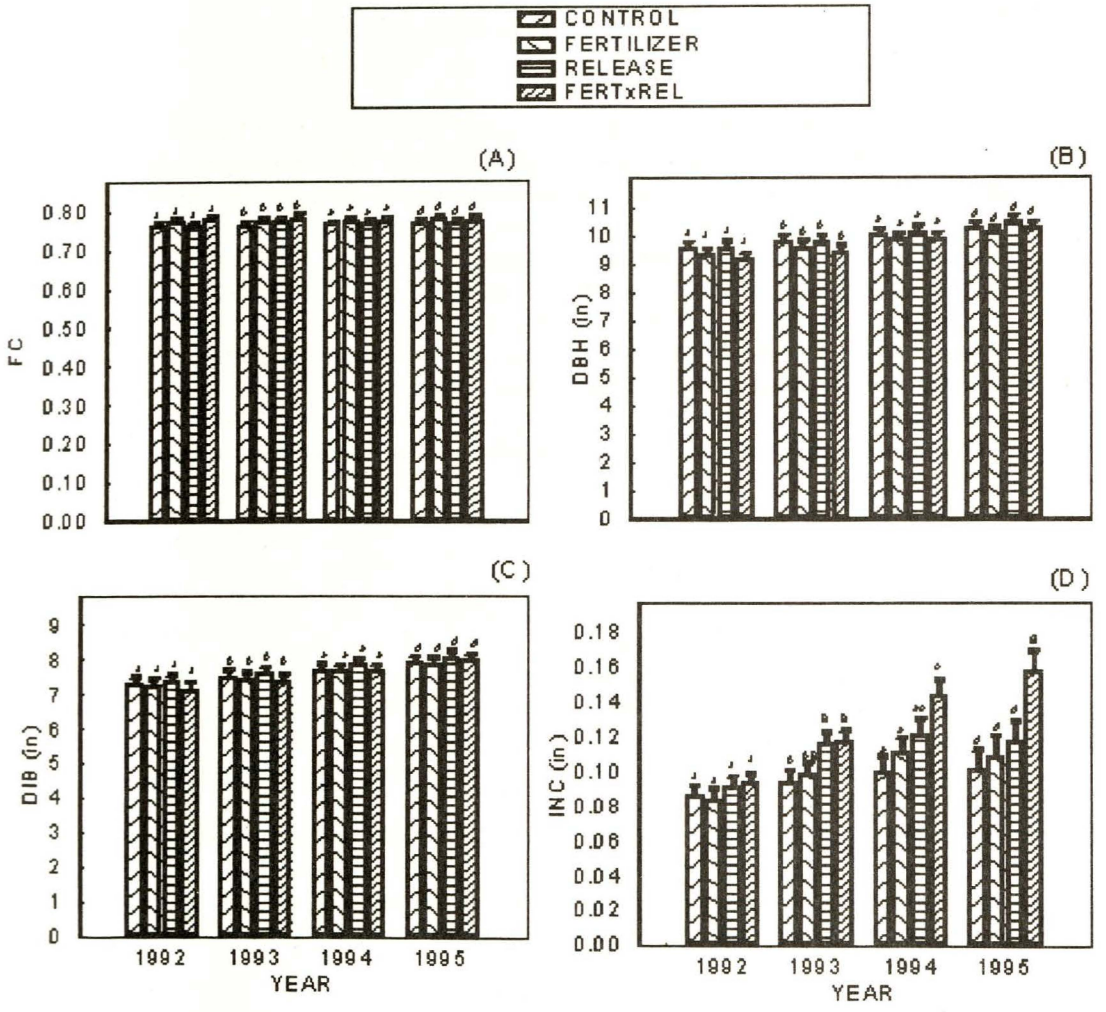


Figure 1. Treatment least square means and 1 SE bars in [A] form class (FC), [B] diameter breast height (DBH), [C] scaling diameter (DIB), and [D] radial increment (INC) of white oak boles at the end of growing season in 1992 (pretreatment year), 1993, 1994, and 1995. Means not labeled by the same letter differ at $P = 0.05$ in a given year.

# WHITE ASH (*FRAXINUS AMERICANA* L.) SURVIVAL AND GROWTH IN UNMANAGED UPLAND FORESTS

Jeffrey S. Ward[1]

Abstract: Crown class and diameter of 704 white ash (*Fraxinus americana* L.) > 0.5 inches dbh have been monitored at 10-yr intervals since 1927. Nominal stand age was 25 years in 1927. Although the density of white ash in the upper canopy declined from 14/acre to 3/acre between stand ages 25 through 85, the proportion of the upper canopy comprised by white ash only fell from 5.2% to 3.6%, respectively. In contrast, both absolute and relative density of lower canopy fell precipitously between stand ages 25 to 85, from 53/acre (4.5%) to 3/acre (0.5%). White ash ingrowth has never exceeded 2.6 trees/acre/decade. Survival and growth of white ash in 25- to 85-yr-old stands were influenced by crown class and disturbance. Mortality was negatively correlated with crown class. Mortality of suppressed trees was 3-5x higher than for trees in the upper canopy. Mortality and diameter growth were independent of tree diameter when crown class effects were removed. Over the 60-yr period, diameter growth for upper canopy, intermediate, and suppressed crown classes averaged 0.8", 0.4", and 0.2"/decade, respectively. Because of low recruitment and lower canopy survival, white ash will likely decline in these forests without periodic disturbances.

## INTRODUCTION

White ash (*Fraxinus americana* L.) is one of the most widespread species in the United States and be found in every state east of the 95th meridian (Schlesinger 1990). There has been concern about a white ash 'decline' and increased mortality since at least the early 1960s (Han and others 1991, Smallidge and others 1991a,b). However, there is very little information on mortality and growth rates of white ash during stand development.

Earlier studies have reported that mortality rates for white ash differ among crown classes. The 10-yr mortality rates ranged from 0.7-3.1% for upper canopy trees, and 23-100% for suppressed trees (Table 1). However, these studies, and studies with mixed crown classes, do not provide information on whether mortality changes during stand development. The rate at which white ash ascends into the upper canopy, or regresses into lower canopy positions is even less well documented.

Fewer studies have documented diameter growth rates (Table 1). Many have small sample sizes. Only one study documented the diameter growth of suppressed white ash (Burns and White 1930). Fastest diameter growth appears to occur between stand ages 25-60, 1.2-3.0 inches/decade. Diameter growth is approximately 1.2 inches/decade in second-growth stands and 1.5 inches/decade in 'old-growth' stands. Unfortunately, these studies do not report basal area growth rates that may be a more appropriate indicator of individual tree vigor.

The objective of this paper is to document development of white ash during natural succession in unmanaged southern New England forests. Individual trees have been inventoried at 10-yr intervals over a 60-yr period in four unmanaged stands in Connecticut. The study includes 704 white ash trees. The long-term monitoring of individual trees permitted an examination of the influence of individual tree characteristics on white ash development. Specifically, this study examined the influence of crown class and diameter class on survival, movement among crown classes, and diameter growth of white ash between stand ages 25-85 yr.

[1]Department of Forestry and Horticulture, The Connecticut Agricultural Experiment Station, 123 Huntington Street, New Haven, CT 06504-1106, Phone: (203) 789-7238, FAX: -7232.

Table 1. Summary of previous research on white ash 10-yr mortality rate (%) and diameter growth (inches/decade) by initial stand age and crown class

| Source | Notes[1] | Initial age[2] | Crown/dbh class[3] | Sample size | Study length | 10-yr rate | |
|---|---|---|---|---|---|---|---|
| | | | | | | Mortality[4] | Dbh[5] |
| Lorimer 1984 | i | 37 | Sup | 109 | 4 | 34.2 | - |
| Lorimer 1984 | i | 42 | Sup | 92 | 14 | 76.38 | - |
| Lorimer 1984 | i | 67 | Sup | 11 | 13 | 100.0 | - |
| Burns and White 1930 | t | SG | Sup | 12 | 11 | 23.0 | 0.3 |
| Conover and Ralston 1959 | t | 11 | Upp | 248 | 16 | 2.8-6.3 | 2.0-3.1 |
| Lorimer 1984 | i | 37 | Upp/Int | 103 | 4 | 0.0 | - |
| Lorimer 1984 | i | 42 | Upp/Int | 103 | 14 | 41.8 | - |
| Zarnovican and Laberge 1994 | i | 52 | Upp | 26 | 4 | - | 1.2 |
| Burns and White 1930 | t | SG | Upp | 71 | 11 | 11.6 | 0.7 |
| Hibbs 1983 | p | 10 | n/a | 1360 | 30 | 45.3 | - |
| Elliott and Swank 1994 | p | 12 | ≥ 2" | 28 | 7 | 29.1 | - |
| Marquis 1969 | e,t | 25 | 3-5" | n/a | 5 | - | 1.5-2.6 |
| Schlesinger 1990 | n/a | 28-38 | n/a | n/a | n/a | - | 1.2-2.0 |
| Schlesinger 1990 | n/a | 20-60 | n/a | n/a | n/a | - | 2.0-3.0 |
| Leak 1970 | i | 40-60 | n/a | n/a | 8-13 | 5.5-7.8 | - |
| Leak 1970 | i | 40-70 | n/a | n/a | 8-13 | 4.4-17.5 | - |
| Wilson 1953 | t | 65 | 5-16" | ~40 | 10 | - | 1.2 |
| Christensen 1977 | p | 120 | n/a | 7 | 22 | 14.2 | - |
| Pallardy and others 1988 | p | SG | < 3.5" | 93 | 14 | 21.4 | - |
| Volk and Fahey 1994 | i | SG | n/a | 194 | 53 | 7.6 | - |
| Fain and others 1994 | p | SG | n/a | >700 | 53 | 2.3 | - |
| Teck and Hilt 1990 | i | n/a | 5-27" | 2028 | 4-17 | 10.2 | - |
| Schlesinger 1990 | n/a | SG | n/a | n/a | n/a | - | 1.2-3.1 |
| Runkle and Yetter 1987 | i | SG | n/a | 3 | 7 | - | 1.3 |
| Teck and Hilt 1991 | i | n/a | 5-27" | 1782 | 4-17 | - | 1.4 |
| Newman and Ebinger 1985 | p | OG | 1-4" | 46 | 18 | 64.9 | - |
| Parker and others 1985 | i | OG | 4-8" | 24 | 50 | 7.8 | 1.68 |
| Parker and others 1985 | i | OG | 8-12" | 45 | 50 | 29.2 | 1.83 |
| Parker and others 1985 | i | OG | 12-16" | 91 | 50 | 22.8 | 1.17 |
| Parker and others 1985 | i | OG | 16-20" | 73 | 50 | 13.7 | 1.48 |
| Parker and others 1985 | i | OG | ≥ 20" | 25 | 50 | 16.7 | 1.37 |
| McGee 1984 | e | OG | > 9" | 275 | 8 | 6.1-7.2 | - |
| Leak 1970 | i | OG | n/a | n/a | 8-13 | 1.8-3.3 | - |

[1] e=estimate, i=individual tree, p-=ate of population decrease, t=thinning study, n/a=not available.

[2] SG=second growth, OG=old growth.

[3] Upp=Upper canopy, Int=intermediate, Sup=suppressed.

[4] Converted to 10-yr rates (%) using $(1-(1-m)^{(10/y)})$ where m=published mortality rates and y=length of study in years.

[5] Diameter growth (inches/decade).

## Study Areas

Study areas are located in the Cabin (40 acres), Cox (50 acres), and Reeves Tracts (40 acres) in Meshomasic State Forest, and the Turkey Hill Tract (80 acres) in Cockaponset State Forest. Most of the land where the plots are located was cleared for farming during the early 1800s and abandoned by the late 1800s. The present forests developed following clearcutting of the second forest for charcoal production around 1900. Prior to fire suppression programs begun during the 1920s, the forests were repeatedly burned to improve grazing and clear brush.

Soils are very stony to extremely stony, fine sandy loams derived from gneiss and schist glacial till with scattered rock outcrops. Topography is gently rolling. Elevations range from 280-650 feet m.s.l. Climatic data are from Hartford, Connecticut, approximately 16 miles northwest of the Meshomasic plots. The area is in the northern temperate climate zone with average annual precipitation of 44 inches (National Oceanic and Atmospheric Administration 1991).

Stand composition and structures are typical of second-growth southern New England forests: upland oak (*Quercus* spp.) in the upper canopy and a lower canopy comprised mostly of maple (*Acer* spp.) and birch (*Betula* spp.). Upland oaks have accounted for more than half the upper canopy basal area since the first inventory (Stephens and Ward 1992). The mean diameter of upper canopy trees increased from 5.1 inches in 1927 to 13.6 inches in 1987 (Ward and Stephens 1994). Study areas were repeatedly defoliated by gypsy moths (*Lymantria dispar*) between 1961-1964, 1971, and 1981. These defoliations greatly increased mortality of upland oak, but not maple and birch (Stephens 1981).

# METHODS

## Field Measurements

The Turkey Hill Tract was established in 1926, and the Cabin, Cox, and Reeves Tracts in 1927. Median age of upper canopy northern red oak (*Quercus rubra*) was 80 years in 1983. The approximate stand age at each measurement, rounded to nearest 5th year, is used rather than the measurement year to facilitate relating results to stages in stand development. Stand ages 25, 35, 55, 65, 75, and 85 will refer to measurements in 1927, 1937, 1957, 1967, 1977, and 1987 respectively.

Trees were mapped to the nearest link (7.92 inches) on transect segments one chain (66 feet) long by 16.5 feet wide. Each transect segment was 1/40 acre. Each tract had 6 to 14 transects and each transect had 10 to 20 transect segments. Transects were spaced at four chain intervals on Cabin, Cox, and Reeves Tracts and five chains intervals on the Turkey Hill Tract. The centerline of each transect was permanently located by a stake and rock cairn at two chain intervals. This study used the 364 transect segments (9.1 acre) that have not been disturbed since 1927. Segments that have been disturbed by partial cutting, road and trail construction, or fire were excluded (3.7 acre).

Transect maps were used to relocate trees in the subsequent inventories of 1937 and 1957. Trees have been relocated since 1967 using the distance along the centerline together with the tree's position right or left of centerline. During each inventory, the species, diameter, and crown class of each tree was recorded. Minimum dbh (diameter at 4.5 feet) was 0.6 inches before 1957; since 1957 the minimum dbh has been 0.5 inches. A metal tape was stretched between stakes to relocate the centerline of each transect segment. Individual trees were matched with data from previous inventories using their location, species, and diameter. New (ingrowth) trees were recorded during each inventory. Inconsistent data (including large ingrowth) were field checked after each inventory to ensure data quality. Crown class designation standards in 1927 and 1937 were based on Fernow (1917), which are nearly identical to those used in later surveys (Smith 1962).

## Data Analysis

Data from the four tracts, which had similar histories, soils, compositions, and climate, were pooled to simplify the analysis. Antecedent crown class is defined as the crown class at the beginning of a period. Four crown class transitions were recognized: ascension (moved into a higher crown class during period), persistence (remained in same crown class), regression (moved into a lower crown class), and mortality.

Crown class transition rates were defined as the proportion of trees in a given antecedent crown class that exhibited crown class ascension, regression, persistence, or mortality during a period. Ten-yr crown class transition matrices were created and squared to obtained 20-yr crown class transition rates for the periods between stand ages 25-35, 55-65, 65-75, and 75-85. This permitted a direct comparison of rates for the 1937-1957 period (stand ages 35-55) with rates for the other periods. Mortality rates were converted to 10-yr mortality rates, as described above, to allow a direct comparison with published rates. Differences in crown class transition rates among periods for each antecedent crown class were tested using procedures in Neter and others (1982, p. 325-329). Differences were considered significant at $P < 0.05$.

A one-way ANOVA with diameter as the dependent and crown class as the categorical variable was used to determine the correlation between crown class and diameter at stand ages 25 and 55 years. To assess the influence of crown class independently of diameter, trees measured at stand age 25 and 55 years were assigned to 3 diameter classes: 0.5-1.9, 2.0-5.9, and >6.0 inches dbh. The Mantel-Haenszel test (SYSTAT 1992) was used to determine whether mortality between stand ages 25-55 and 55-85 was independent of canopy class (upper vs. lower canopy) when stratified by diameter class, and diameter class when stratified by crown class. Differences were considered significant at $P < 0.05$.

Analysis of variance was used to examine influence of crown class and stand age (fixed effects) on diameter growth with initial diameter as a covariates on diameter growth. Tukey's HSD test was used to determine whether diameter growth differed among crown and diameter classes within and among intersurvey intervals. Differences were considered significant at $P < 0.05$.

# RESULTS

White ash density steadily declined over the past 60 years, from 67 stems/acre at stand age 25 to 6 stems/acre at stand age 85 (Table 2). Concurrently, dispersal of white ash decreased from 40% of transect segments at stand age 25 to 11% at stand age 85. Most of the decline has occurred in the suppressed and intermediate crown classes. Lower canopy density decreased 94% between stand ages 25 to 85. Although the density of upper canopy white ash decreased 76% between stand ages 25-85, the proportion of the upper canopy comprised of white ash only decreased from 5.2% to 2.6%. White ash ingrowth has averaged less than 2.5 trees/acre/decade.

## Crown Class Transitions

Survival and crown class transitions of white ash were not independent of antecedent crown class. The distribution of trees among crown classes at the end of a period (including mortality) was not independent of antecedent crown class for each of the five periods (stand ages 25-35, 35-55, 55-65, 65-75, and 75-85). Chi-square values ($\chi^2$) for all periods were highly significant ($p < 0.001$).

Table 2. Stand density (stems/acre), mean dbh (inches), ingrowth (stems/acre/decade), and frequency of white ash in Connecticut by crown class and stand age

| Crown class | Stand age (yr) | | | | | |
|---|---|---|---|---|---|---|
| | 25 | 35 | 55 | 65 | 75 | 85 |
| | Density (stems/acre) | | | | | |
| Upper canopy | 11.3 | 6.5 | 4.3 | 5.2 | 3.8 | 2.7 |
| Intermediate | 14.8 | 7.9 | 2.7 | 2.4 | 1.1 | 1.0 |
| Suppressed | 40.5 | 28.8 | 12.1 | 3.8 | 2.6 | 2.0 |
| Combined | 66.7 | 43.2 | 19.1 | 11.4 | 7.6 | 5.7 |
| | Mean diameter (inches) | | | | | |
| Upper canopy | 4.5 | 6.3 | 8.0 | 8.2 | 9.5 | 11.0 |
| Intermediate | 2.4 | 3.9 | 5.3 | 5.2 | 6.1 | 7.0 |
| Suppressed | 1.2 | 1.9 | 2.0 | 1.3 | 1.4 | 1.5 |
| Ingrowth | - | 2.3 | 2.6 | 0.9 | 1.3 | 0.9 |
| Frequency [1] | 39.8 | 34.1 | 22.3 | 16.2 | 12.6 | 11.0 |

[1] Percent of transect segments (1/40 acre) with at least one white ash.

Crown class ascension rates were not stationary from stand ages 25 through 85 (Table 3). The crown class ascension rate of intermediate trees between stand ages 55-65 was 3-21 times higher than earlier or later periods. This rate was significantly higher than for all other periods except stand ages 75-85. The crown class ascension rate of suppressed trees also peaked between stand ages 55-65. This rate was 4-9 times higher for other periods and significantly higher than for earlier periods. Intermediate trees moved into the upper canopy at higher rates than suppressed trees moved into the intermediate crown class during all periods (Table 3).

Two general pattern of crown class persistence were noted (Table 3). Persistence of white ash in upper canopy and the intermediate crown class peaked between stand ages 55-65. Nearly 96% of white ash in the upper canopy at stand age 55 remained in the upper canopy through stand age 65. The opposite pattern was observed for trees in the suppressed crown class. Crown class persistence between stand ages 55-65 was significantly lower than for the previous 30 years because crown class ascension greatly increased. Persistence of intermediate trees was lower than for upper canopy trees for all periods, and lower than for suppressed trees for all periods except between stand ages 55-65.

Temporal patterns of crown class regression were similar for upper canopy and intermediate white ash (Table 3). Crown class regression rates were higher between stand ages 25-55 than for the next 30 years. Regression rates were at their lowest between stand ages 55-65, the period of defoliation and drought. Regression rates began to rise after stand age 65, but still were much lower than prior to stand age 55. Crown class regression rates of intermediate trees was lower than for upper canopy trees for all five periods.

Table 3. 20-yr crown class transition rates (%) for white ash in Connecticut by antecedent crown classes among stand age intervals. Column values with same letter for a given crown class were not significantly different at $p < 0.05$.

| Crown class at beginning of period | Period (stand age) | 20-yr crown class transition rate (%) | | | | | |
|---|---|---|---|---|---|---|---|
| | | Ascension | | Persistence | | Regression | |
| Upper canopy | 25-35 | - | | 28.6 | a | 56.7 | c |
| Upper canopy | 35-55 | - | | 52.5 | b | 28.8 | b |
| Upper canopy | 55-65 | - | | 95.9 | c | 3.3 | a |
| Upper canopy | 65-75 | - | | 49.9 | b | 10.5 | ab |
| Upper canopy | 75-85 | - | | 47.2 | ab | 19.1 | b |
| Intermediate | 25-35 | 2.2 | a | 6.3 | a | 36.5 | b |
| Intermediate | 35-55 | 6.9 | ab | 11.1 | a | 22.2 | b |
| Intermediate | 55-65 | 46.6 | c | 11.2 | a | 0.0 | a |
| Intermediate | 65-75 | 3.6 | ab | 1.9 | a | 6.9 | ab |
| Intermediate | 75-85 | 15.1 | bc | 5.8 | a | 8.7 | b |
| Suppressed | 25-35 | 1.1 | a | 18.59 | b | - | |
| Suppressed | 35-55 | 1.1 | a | 17.56 | b | - | |
| Suppressed | 55-65 | 10.2 | b | 6.48 | a | - | |
| Suppressed | 65-75 | 1.2 | ab | 8.68 | ab | - | |
| Suppressed | 75-85 | 2.6 | ab | 11.94 | ab | - | |

Mortality

The mortality rate of upper canopy white ash has always been significantly lower for intermediate and suppressed trees (Table 4). The rate for intermediate trees was significantly higher than for suppressed trees prior to stand age 65. Two general temporal patterns were noted for mortality when trees were segregated by crown class. Mortality rates of suppressed white ash increased slightly, but not significantly, between stand ages 25-85 (Table 4). There was also a long term trend of increasing mortality rates for both intermediate and upper canopy trees. Mortality rates for both crown classes were significantly higher between stand ages 75-85. In contrast with the pattern for suppressed white ash, mortality rates for the higher crown classes decreased between stand ages 55-65 (Table 4).

Mortality decreased with increasing tree diameter (Table 4). Crown class and diameter were highly correlated at stand age 25 ($r^2 = 0.498$, $p<0.001$). White ash mortality between stand ages 25-55 was independent of diameter class when crown class effects were removed (Mantel-Haenszel $\chi^2 = 3.04$, $p<0.081$), but not independent of canopy class (upper vs. lower canopy) when diameter class effects were removed (Mantel-Haenszel $\chi^2 = 44.95$, $p<0.001$). Therefore, mortality estimates based on diameter were improved by also using canopy class, but mortality estimates based crown class alone were not improved by also using diameter. Similar relationships were noted between stand ages 55-85. Crown class and diameter were highly correlated at stand age 55 ($r^2 = 0.617$, $p<0.001$). Mortality of white ash between stand ages 55-85 was independent of diameter class when crown class effects were removed (Mantel-Haenszel $\chi^2 = 0.22$, $p<0.638$), but not independent of canopy class when diameter class effects were removed (Mantel-Haenszel $\chi^2 = 25.12$, $p<0.008$).

Table 4. Significance of 10-yr mortality rates (%) for white ash in Connecticut among periods (stand age intervals) by crown and diameter classes.

| | Period (stand age) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 25-35 | | 35-55 | | 55-65 | | 65-75 | | 75-85 | |
| | Significance among periods[1] | | | | | | | | | |
| Upper canopy | 7.7 | b | 9.8 | bc | 0.4 | a | 21.1 | d | 18.6 | cd |
| Intermediate | 32.9 | a | 36.5 | a | 24.0 | a | 64.8 | b | 45.6 | ab |
| Suppressed | 55.7 | a | 56.8 | a | 59.2 | a | 68.6 | a | 61.8 | a |
| > 6.0" dbh | 4.2 | a | 14.4 | ab | 4.8 | a | 26.1 | b | 18.4 | b |
| 2.0-5.9" dbh | 24.2 | a | 45.0 | b | 52.6 | b | 58.0 | b | 50.0 | b |
| 0.5-1.9" dbh | 58.2 | a | 56.8 | a | 64.3 | a | 63.0 | a | 58.1 | a |
| | Significance among classes[2] | | | | | | | | | |
| Upper canopy | a | | a | | a | | a | | a | |
| Intermediate | b | | b | | b | | b | | b | |
| Suppressed | c | | c | | c | | b | | b | |
| > 6.0" dbh | a | | a | | a | | a | | a | |
| 2.0-5.9" dbh | b | | b | | b | | b | | b | |
| 0.5-1.9" dbh | c | | c | | b | | b | | b | |

[1] row values with the same letter were not significantly different at $p \leq 0.05$.

[2] column values with the same letter were not significantly different at $p \leq 0.05$ than between stand ages 25-35.

Diameter Growth

Diameter growth (inches/decade) was significantly different among crown classes ($F=65.3$, d.f.=2, $p<0.001$), and diameter classes ($F=29.1$, d.f.=2, $p<0.001$). Mean diameter growth was significantly greater for trees in the upper canopy than for suppressed trees, 0.78 and 0.25 inches/decade, respectively. Mean diameter growth was significantly greater for larger white ash (> 6" dbh) than for smaller trees (0.5-1.9" dbh), 0.75 and 0.31 inches/decade, respectively. After stand age 65 diameter growth was independent of diameter, but not crown class. Within a crown or diameter class, diameter growth was constant between stand ages 25-85 (Table 5).

## DISCUSSION

The crown class of an individual white ash had a strong influence on future survival and development of that tree. For example, while 25.2% of upper canopy trees in the 25-yr-old stand survived through stand age 85, and 17.5% remained in upper canopy positions, only 2.2% of intermediate trees survived between stand ages 25 to 85. Therefore, number of white ash that can be expected in future stands depends on both the number of trees and their distribution among the crown classes. Thirty-year crown class transition values (stand ages 25-55) were calculated from the data to facilitate comparison with earlier reports on other species. The results show that 332 upper canopy or 6,757 intermediate white ash at stand age 25 are necessary to obtain 100 upper canopy trees at stand age 55. This compares with 550 and 1,730 for yellow birch (*Betula alleghaniensis*), 440 and 3,570 for red maple (*Acer rubrum*), and 170 and 1,200 for northern red oak (Ward and Stephens 1993, 1994). White ash is more likely to remain in the upper canopy on these sites than red maple or yellow birch, but not northern red oak. However, white ash is much less likely to ascend from the intermediate crown class into upper canopy positions than the other co-occurring species.

Table 5. Significance of diameter growth rates (inches/decade) for white ash in Connecticut among periods (stand age intervals) by crown and diameter classes

| | Period (stand age) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 25-35 | | 35-55 | | 55-65 | | 65-75 | | 75-85 | |
| | | | | | Significance among periods[1] | | | | | |
| Upper canopy | 0.88 | a | 0.62 | a | 0.77 | a | 0.71 | a | 0.88 | a |
| Intermediate | 0.44 | a | 0.32 | a | 0.49 | a | 0.23 | a | 0.32 | a |
| Suppressed | 0.27 | a | 0.22 | a | 0.21 | a | 0.25 | a | 0.22 | a |
| > 6.0" dbh | 1.01 | a | 0.74 | a | 0.63 | a | 0.73 | a | 0.85 | a |
| 2.0-5.9" dbh | 0.68 | a | 0.36 | b | 0.51 | ab | 0.39 | ab | 0.33 | ab |
| 0.5-1.9" dbh | 0.32 | a | 0.29 | a | 0.20 | a | 0.28 | a | 0.30 | a |
| | | | | | Significance among classes[2] | | | | | |
| Upper canopy | a | | a | | a | | a | | a | |
| Intermediate | b | | b | | ab | | a | | ab | |
| Suppressed | b | | b | | b | | a | | b | |
| > 6.0" dbh | a | | a | | a | | a | | a | |
| 2.0-5.9" dbh | a | | b | | ab | | a | | a | |
| 0.5-1.9" dbh | b | | b | | b | | a | | a | |

[1] row values with the same letter were not significantly different at $p \leq 0.05$.
[2] column values with the same letter were not significantly different at $p \leq 0.05$.

A significant advantage for white ash that ascend to upper canopy positions is reduced mortality rates relative to trees in subordinate canopy positions (Table 4). This has been true for the entire 60 years of this study and is consistent with earlier reports (Lorimer 1981, 1984). The mortality rates observed in this study are comparable with those in earlier reports (Table 1). Upper canopy mortality rates of this study (1-21%/decade) were somewhat higher than published rates (3-12%/decade). Mortality rates of suppressed trees were more consistent than earlier studies (56-69%/decade vs. 23-100%/decade, respectively).

White ash is highly resistant to gypsy moth defoliations (Herrick and Gansner 1987, Liebhold and others 1995). During the period of repeated defoliations, between stand ages 55-65, mortality rates fell sharply for intermediate and upper canopy white ash (Table 4). This is in marked contrast to the sharply increased mortality of upper canopy northern red oak during the same period (Ward and Stephens 1994). The sharp increase in crown class ascension rates during the decade of defoliation (Table 3) indicates that lower canopy white ash respond rapidly to release. Crown class regression rates increased following subsidence of defoliation intensity, but at lower rates than before stand age 55. Considering the decreased mortality and crown class ascension rates, it is surprising that there was not a concurrent increase in diameter growth (Table 5).

The very low density of white ash ingrowth (Table 2) fits a pattern gleaned from earlier research. Highest levels of white ash regeneration have been reported in the Midwest (Den Uyl 1961, Sander and Clark 1971, Parker and Leopold 1983, Boerner and Cho 1987, Martin and Hix 1988, Hix and Lorimer 1991, Roberts 1992). Relative to the Midwest, white ash regeneration declines to the west (Pallardy and others 1988, Shotola and others 1992), northeast (Hannah 1991, Smallidge and others 1991, Fain and others 1994, Volk and Fahey 1994), and southeast (Christensen 1977, Della-Bianca 1983, Clebsch and Busing 1989, Clinton and Boring 1994). Without a series of disturbance events that dramatically increases white ash regeneration density, white ash will gradually disappear from the forests in this study as mature trees die.

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# MODELING FOREST LANDSCAPE CHANGE IN THE OZARKS: GUIDING PRINCIPLES AND PRELIMINARY IMPLEMENTATION

Stephen R. Shifley[1], Frank R. Thompson III[1], David R. Larsen[2], and David J. Mladenoff[3]

Abstract: Although research and management approaches for ecosystem management have been troublesome to define, we believe a number of guidelines can be used to focus research. These include: (1) synthesize the extensive information describing hardwood ecosystem response to disturbance, (2) ensure that syntheses cross traditional boundaries between disciplines and resources, (3) develop principles that operate at the broad scale and avoid getting mired in excessive detail, (4) incorporate a modeling framework to organize information and relationships, and (5) focus on mechanisms of disturbance. Disturbance is a normal, ongoing process in Central Hardwood ecosystems. It takes many forms, but among the most extensive are weather damage, fire, and harvest and regeneration practices. Much is already known about how Central Hardwood forests respond to disturbance, but this knowledge is too poorly organized to allow us to explore the impacts of disturbance at the landscape scale. We present a structured framework for organizing and modeling ecosystem attributes (and their change over time) in Central Hardwood landscapes. The model is based on the premise that forest vegetation communities follow patterns of development that, although highly variable, have central tendencies that can be predicted through time with reasonable accuracy. After a vegetative community is altered by disturbance, it eventually resumes a characteristic (often highly stochastic) pattern of development. Although vegetation characteristics are dynamic in time, they are spatially anchored. Knowledge of the vegetation condition at a given place and time allows us to associate other ecosystem attributes that can be estimated from vegetation conditions (e.g., economic value, wildlife habitat suitability, scenic beauty). For a large forested landscape, a suitable minimum scale of analysis is the stand, typically about 5 ha in size with boundaries that often coincide with physiographic features. Patterns at this scale can be aggregated and analyzed to evaluate ecosystem attributes that operate at larger scales. Tools to facilitate this procedure are already available or can be adapted to the task. We recalibrated the LANDIS model for the species and conditions in the Missouri Ozarks and used that model to simulate landscape change through time. We demonstrate the model showing simulated patterns of disturbance and vegetation response for a small Ozark forest landscape.

## INTRODUCTION

In the past several years, increasing emphasis has been placed on understanding the ecological implications of management activities at landscape scales. This movement has been evident in public land management agencies, and the USDA Forest Service has taken a particularly active role. This shift in emphasis from managing individual stands to managing mosaics of forest stands for multiple resources within a landscape has met with some confusion both in terminology and in implementation. "New Perspectives" and "Ecosystem Management" have been two of the widely used monikers to describe this initiative. In the Forest Service, some initiatives appear destined to have a short life span or limited scope from the outset; others catch hold and lead to fundamental changes. From our view, the ecosystem management initiative falls into the latter group.

[1] Principal Biometrician and Project Leader, respectively. North Central Forest Experiment Station, USDA Forest Service, 1-26 Agriculture Building, University of Missouri, Columbia, MO 65211.

[2] Assistant Professor, School of Natural Resources, 1-26 Agriculture Building, University of Missouri, Columbia, MO 65211.

[3] Assistant Professor, Department of Natural Resources, University of Wisconsin, 1630 Linden Drive, Madison, WI 53706.

Nevertheless, the whole concept of ecosystem management has resulted in considerable head-scratching on our own part as we considered the following questions:

What does ecosystem management mean in the Central Hardwood Forest complex?
How does it differ from the management practices that are already in place, many of which already embrace some of the concepts embodied in ecosystem management?
What type of research is necessary to address issues surrounding ecosystem management?
What, if anything, can we contribute to this initiative?

In this paper we address these questions and several others as we describe our path to a better understanding of ecosystem management and ecosystem management research in Central Hardwood forests.

## SEEKING A DEFINITION OF ECOSYSTEM MANAGEMENT

The initial problem with ecosystem management is one of definition. Arriving at a working definition is essential, but all definitions of ecosystem management are vague or at least nonspecific. A common refrain is "I can't describe ecosystem management, but I know what it is when I see it."

One formal definition of ecosystem management from the USDA Forest Service (1994) is the following:

> *Ecosystem management is not an end in itself, but a means to achieve sustainable conditions and provide wildlife and fish habitat, outdoor recreation, wilderness, water, wood, mineral resources, and forage for society while retaining the esthetic, historic, and spiritual qualities of the land.*

Note that the initial focus of this definition is on what ecosystem management is *not* rather than what it is. This simply underscores the fact that it is easier to identify those things that clearly are not representative of ecosystem management than to succinctly define those that are.

Another Forest Service definition of ecosystem management is that of Bartuska (1993):

> *Ecosystem management means using an ecological approach to achieve the multiple-use management of national forests and grasslands by blending the needs of people and environmental values in such a way that national forests and grasslands represent diverse, healthy, productive, and sustainable ecosystems.*

A third definition is that of Thomas (1996):

> *Simply stated, ecosystem management is the integration of ecological, economic, and social factors in order to maintain and enhance the quality of the environment to meet current and future needs. It is a holistic approach to natural resource management. To use it, we must step back from what foresters call the stand level and focus on the broader landscape.*

He notes that ecosystem management is in keeping with the multiple-use mandate of the Forest Service, but is more inclusive in terms of characteristics evaluated, larger spatial scales, and longer time frames.

Despite some inevitable degree of ambiguity in the definition of ecosystem management, several characteristics are consistently associated with the concept: (1) ecosystem management maintains diverse, healthy forest ecosystems; (2) it stresses sustainability; (3) it blends needs of people and environmental values by including people as an integral part of the ecosystem; (4) it considers effects at multiple spatial and temporal scales; (5) it addresses uncertainty and risk; (6) it emphasizes monitoring and evaluation; and (7) it employs adaptive management techniques.

"It's having and using a lot more information about much larger areas"[4] is among the more practical definitions of ecosystem management that we have heard. Although this definition is decidedly nontechnical and nonspecific, it captures the essential elements of large landscapes and multiple values.

[4] From Bill Sexton, Deputy Director of Ecosystem Management, USDA Forest Service, Washington, DC.

## RESEARCH OPPORTUNITIES

We could make excellent progress with ecosystem management research in the Central Hardwood region if we could select an ecosystem, say a large watershed, and answer the following questions:

— If we apply a particular management practice, what is the impact on ecosystem integrity or health?
— If we apply a particular management practice, what is the impact on ecosystem sustainability?
— If we apply a particular management practice, what is the impact on ecosystem productivity?

Unfortunately these questions hide other more fundamental questions. Namely:

— What is ecosystem integrity or health, and how do we measure it?
— What is sustainability, and how do we measure it?
— What is productivity, and how do we measure it?

Faced with this degree of intractability in defining research issues, we had to narrow our consideration to research that is both possible and practical. In our discussions of ecosystem management research, we have concluded that synthesis of existing information is essential. We will never have perfect knowledge about ecosystem composition, structure, and function. Consequently, there is no need to wait for the completion of some definitive study before we start drawing conclusions relevant to ecosystem management. In fact, we already have vast knowledge about many aspects of forest ecosystems. For example, literally thousands of publications deal with oak silviculture. More than 250 publications address growth and yield in the Central Hardwood forest. The base of published literature for eastern old-growth forests exceeds 700 articles (Nowacki and Trianosky 1993); for savannas, the body of published literature consists of more than 450 items[5]. Paul Johnson[6] has termed this wealth of knowledge "an embarrassment of riches" relative to hardwood forest management. Despite this abundance of literature, what we collectively understand about hardwood forests is poorly organized for addressing the issues of ecosystem management in the Central Hardwood region. Specifically, much of what we understand about forest vegetation dynamics is limited in geographic scope and in spatial scale (e.g., to stand-level relevance). Linkages across resources or disciplines such as silviculture, economics, growth and yield, recreation, scenic preference, wildlife, and water quality are generally weak. Linkages across spatial scales are typically even weaker. Linkages from processes that operate at large scales (e.g., large mammal movements, fire or wind disturbance, regional economic health, water quality protection) to those that operate at relatively small scales (e.g., the vast majority of contemporary silvicultural practices) are central to ecosystem management.

Several principles have guided and prioritized our venture in ecosystem management research. First, for reasons outlined above, we believe that one of the most productive ways to begin in the Midwest is to synthesize what is already known about these forests. This includes synthesis for multiple resources: vegetation, wildlife, aesthetic quality, economic value, recreation, and water quality.

Second, syntheses must cross the traditional functional lines of silviculture, wildlife, economics, recreation, and other relevant disciplines. The scope of this work requires a cooperative effort.

At a fine level of detail, this synthesis has the potential to become overwhelming. Consequently, a third guiding principle is to seek the big picture at the landscape scale before getting mired in site-specific details. A landscape is " a heterogeneous land area composed of a cluster of interacting ecosystems" (Crow 1991). Obviously, there is considerable latitude in defining the spatial extent of a landscape, but landscapes encompassing tens to hundreds of square kilometers are typical. In the Midwest, this is an appropriate scale for integrated, multiple-resource forest management and for assessment of cumulative effects (both spatially and temporally).

Fourth, some sort of a modeling framework is essential. There are simply too many components to handle in any other manner. Modeling has the added advantage (if carefully designed) of allowing us to begin by formalizing general principles and processes and then adding site-specific details as they become necessary to address specific management issues.

[5] Personal communication, Alan Rebertus, School of Natural Resources, University of Missouri, Columbia, MO.
[6] Pincipal Silviculturist, North Central Forest Experiment Station, USDA Forest Service, Columbia, MO.

Finally, midwestern hardwood ecosystems have been continuously shaped by disturbance. Disturbance is a normal ongoing process. Consequently, predicting changes in ecosystem attributes in response to disturbances is a necessary prerequisite for resource management. Generically, a disturbance is any temporally discrete event that disrupts the structure, resource availability, or physical environment of a population, community, or ecosystem (White and Pickett 1985). Three disturbance factors that have great historical and contemporary importance in the Central Hardwood region are wind, fire, and timber harvest. Other factors such as insects and disease are of great importance in certain localities. For a specific, limited set of disturbance processes, we must answer the question for a real ecosystem, "How will this ecosystem respond to any given sequence of disturbance events?" More specifically, we must focus on those ecosystem attributes for which we have some knowledge and that are relevant to management decisionmaking. For a given ecosystem, this leads to the researchable questions: "How do attributes of vegetation composition, vegetation structure, wildlife habitat quality, economic value or aesthetic quality change over time in response to disturbance?"; "How are those changes spatially arranged on the landscape?"; and "What are the implications of those changes and tradeoffs that are relevant to management decisions?"

## SCALE

All definitions of ecosystem management incorporate the concept of evaluating impacts at multiple scales. But what scales are appropriate? By definition an ecosystem can be as small as a drop of water or as large as the biosphere. Our ability to develop linkages across this range of scales is nonexistent in the foreseeable future, so what scales are essential? Silvicultural treatments in midwestern forests (and hence harvest disturbances) are prescribed for individual stands (roughly 5 ha). Thus, the finest scale must be at least this small. On the other hand, ecosystem management incorporates landscape-scale considerations.

From a practical perspective, relevant landscape sizes are typically on the order of tens to hundreds of square kilometers. The largest scales are dictated by a combination of administrative, ecological, and technical factors. National Forests in the Midwest are organized into ranger districts or administrative units that may be as large as 100,000 ha. From an ecological perspective, large-scale management planning may be best conducted within ecologically similar units termed subsections (McNab and Avers 1994, USDA Forest Service 1981). These may be hundreds of square kilometers in size. Additionally, an analysis unit ideally should be several times larger than the home range of the important animals that live in the unit.

There are two major technical limitations at the large end of the scale. One is the capacity of computing technology to model, analyze, and display information on large tracts. Although often constraining at first glance, computational capacity is almost never a limitation of consequence. In the length of time it takes to develop and test a landscape-level model, limitations in computing capacity that appear constraining at the start of the project are inevitably eliminated by subsequent technological advances in hardware and software. The technical limitation of greater consequence is finding a landscape-level data base suitable for model development and application. If, for example, projections of change across a large landscape are to be based on current vegetation composition and structure, site quality, topography, and historical patterns of wind, fire, and harvest disturbances, then all those attributes must be known or predictable across the entire landscape. There are few places in the Central Hardwood region where such detailed, spatially explicit databases exist.

Fortunately, there are two such data bases in the Ozarks of southeastern Missouri. They are associated with the Mark Twain National Forest (approximately 105,000 ha) and the Missouri Ozark Forest Ecosystem Project (3,725 ha) (Brookshire and Hauser 1993). These areas have detailed spatial data with layers describing topography, ecological land type, and current vegetation spatially referenced in a geographic information system (GIS). In addition, the ongoing Ozark/Quachita Highlands Assessment will provide more extensive information at an even larger scale.[7] This region is characterized by a predominantly forested landscape that is interrupted by clearings for pasture in the major stream and river bottoms (Figure 1). Annual harvest is approximately 40% of growth and less than 1% of the standing inventory (Leatherberry 1990). Harvest is principally by the selection method (either by silvicultural group-selection or individual-tree-selection or by "logger's choice" selection).

[7] Cooperative project by Forest Service Regions 8 and 9 and the North Central and Southern Forest Experiment Stations.

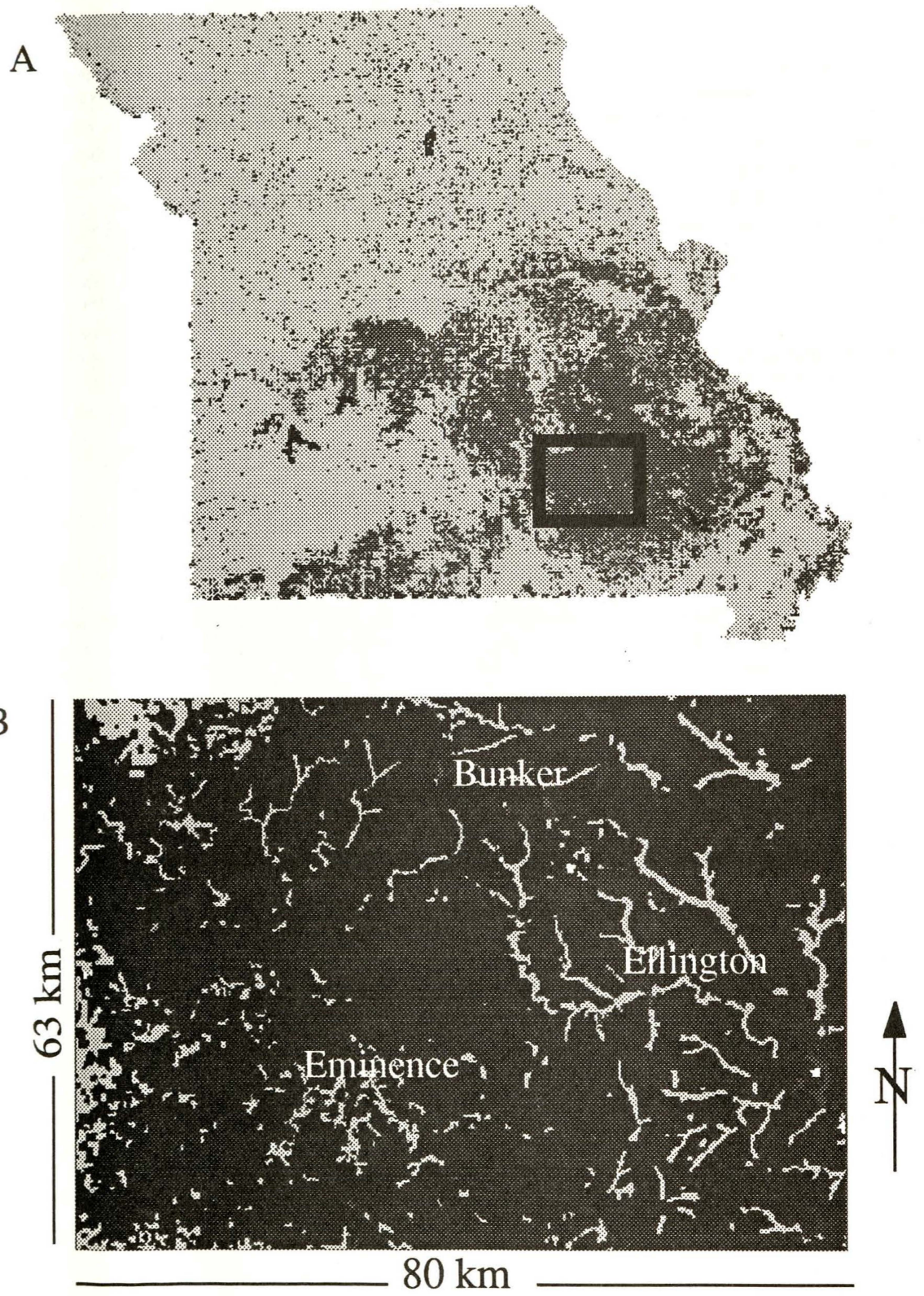


Figure 1. (A) Missouri forest cover showing general location of mapped forest landscapes amenable to detailed analysis and simulation. (B) Characteristic view of vegetation cover in southeast Missouri Ozarks. The area is predominantly forested (dark shading) with interspersed agricultural lands (light shading) that are virtually all pasture.

Fire is a dominant factor shaping the vegetation of the Ozarks, but the nature of fire disturbance has changed over time. Before European settlement (1580-1820) the mean fire-free interval was approximately 12 to 18 years. Between 1821 and 1940 the mean fire-free interval had decreased to 4 years (Guyette 1995). In the years since then, the major impact of fire has come from its exclusion rather than its presence. Wind, also, is a disturbance factor that shapes these forests through the creation of both small and large openings in the forest canopy. Mean return intervals for non-catastrophic wind events in the Ozarks are on the order of 260 years[8], but tornadoes (26 per year in Missouri) add to wind disturbance.

## PREDICTING LANDSCAPE CHANGE THROUGH TIME

Our goal in this research is to predict change in landscapes at spatial scales as large as a few hundred thousand hectares while tracking details for individual, mapped stands as small as a few hectares. From this we can estimate and contrast changes in an array of ecosystem attributes that result from alternative disturbance (i.e., harvest, fires, wind) regimes.

We make the following assumptions in our model:

1. Changes in vegetation composition and structure follow predictable patterns of development and respond in predictable ways to disturbance.
2. The landscape can be divided into ecologically similar units that have similar patterns of vegetation development and similar disturbances.
3. If we know (or have predicted) vegetation composition and structure for a mosaic of stands at a future point in time, we can say significant things about other ecosystem properties including those that operate at larger spatial scales (e.g., wildlife populations, economic value, aesthetic quality, product yields, landscape structure, biological diversity).

Fortunately, the implementation of this process in the Missouri Ozarks can draw on a number of existing technologies and research tools. The general idea of projecting landscape change has been prevalent since GIS became a practical tool for forest management in the late 1980's. Since that time the practicality of landscape scale simulation has increased markedly as spatial analysis software has improved, computing power has increased, and characteristics of forest landscapes have been incorporated into spatial (GIS) and information databases.

Also, software to simulate vegetation change in response to natural disturbance patterns has been developed for other regions (e.g., Botkin 1993, Jones and others 1995, Mladenoff and others 1996). The LANDIS model (He and others 1996, Mladenoff and others 1996) is well-suited for adaptation to the Missouri Ozarks. LANDIS characterizes a landscape as a mosaic of square "sites" or pixels. Each site is geographically referenced (mapped) and identified with a specific ecological land unit. Vegetation at each site is represented by a dynamic list of tree species cohorts present by age class. Stochastic equations predict changes over time in the number of cohorts by species and age class at a given site. Predicted changes at a given site respond to the presence or absence of wind and fire disturbances. Wind and fire disturbances are represented as spatially explicit stochastic events on the landscape. The frequency and size of those disturbance events can be altered to investigate different ecosystem responses. We have calibrated LANDIS for the 16 most common tree species and 7 generalized ecological land types adapted from the Forest Service ecological land classification guide for the Missouri Ozarks (USDA Forest Service 1981). We are in the process of developing techniques that will realistically simulate the disturbance of harvesting and that will incorporate linkages from vegetation at each site to other ecosystem attributes (e.g., wildlife habitat, regional economic value, aesthetic quality, product yields, landscape diversity, biological diversity).

Several examples of implementation of LANDIS in the Missouri Ozarks illustrate our approach. We started with a mapped landscape showing ecological land units (Figure 2). This landscape consists of 839 ha (Compartments 7 and 8) of the Missouri Ozark Forest Ecosystem Project (Brookshire and Hauser 1993). This landscape was subdivided into 9,323 30 m-by-30 m sites (or pixels) for simulation with LANDIS (He and others 1996, Mladenoff and

[8] Personal communication, Alan Rebertus, School of Natural Resources, University of Missouri, Columbia, MO.

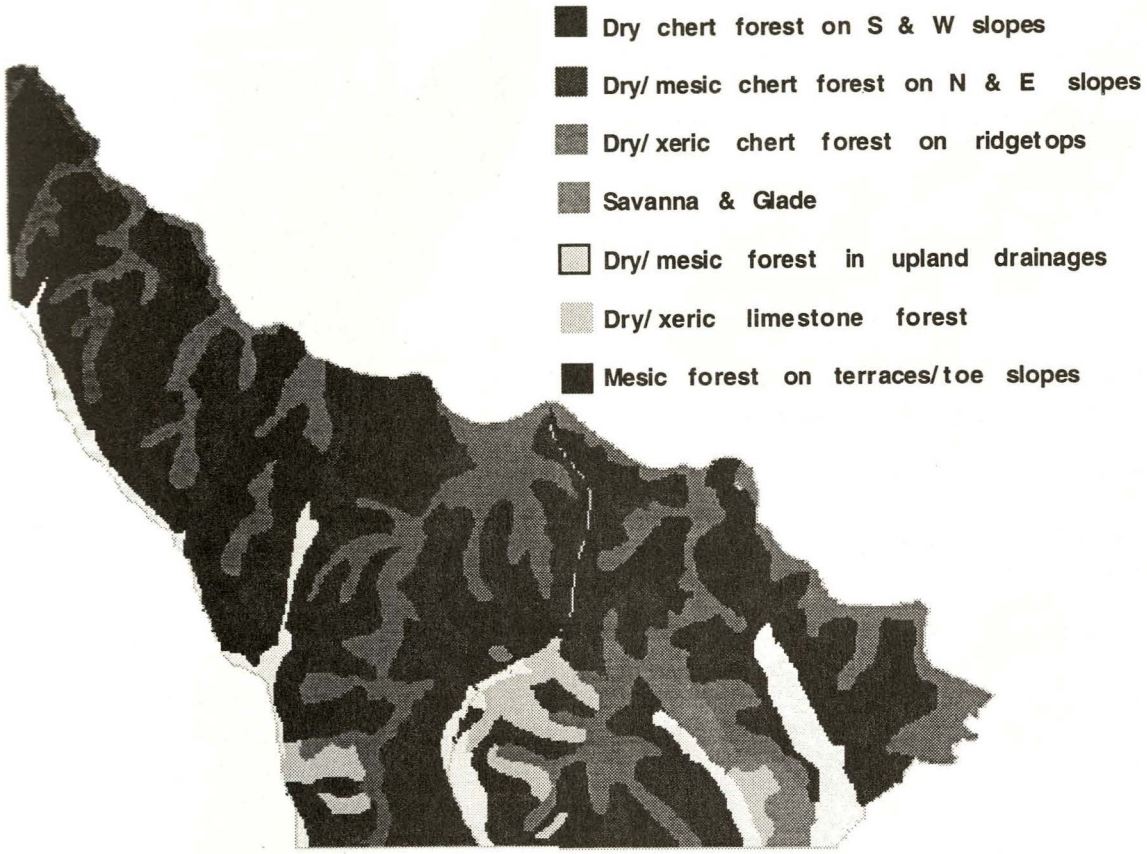
Dry chert forest on S & W slopes
Dry/ mesic chert forest on N & E slopes
Dry/ xeric chert forest on ridgetops
Savanna & Glade
Dry/ mesic forest in upland drainages
Dry/ xeric limestone forest
Mesic forest on terraces/toe slopes

Figure 2. Missouri Ozark Forest Ecosystem Project (MOFEP) Compartments 7 and 8, Carter County, Missouri. Ecological land units are based on USDA Forest Service (1981).

others 1996). We simulated change in vegetation composition and age structure and the effects of fire and wind disturbance on patterns of development.

We compared fire disturbance events at two frequencies (Figure 3). The low frequency fire regime had a mean return interval of approximately 100 years. The high frequency regime had a mean return interval of 4 years, approximating fire frequency in the Ozarks from 1821 to 1940 (Guyette 1995). We plotted cumulative wind disturbance over a century (Figure 4) as the summation of simulated wind events with random spatial coordinates and with a size distribution of events corresponding to estimates of large-scale wind events in southeastern Missouri. The results in Figure 4 are based on a mean event size of 5 ha and a mean return interval of 260 years, values that are characteristic for the Missouri Ozarks.

The tree species and age classes associated with each site changed in response to the presence or absence of disturbance events. We displayed species information in several different ways. The maximum age of trees by site can be displayed at any point in time (Figure 5). We also mapped the distribution of dominant species (Figure 6). Although Figures 5 and 6 are illustrations for one point in time, the values at each site change each decade in the course of a simulation and can be thought of as a moving picture through time. The presence or absence of individual species can also be tracked through time over the course of a simulation. For instance, we displayed the presence and absence of white oak and red maple on the simulated landscape at two time periods 50 years apart (Figure 7).

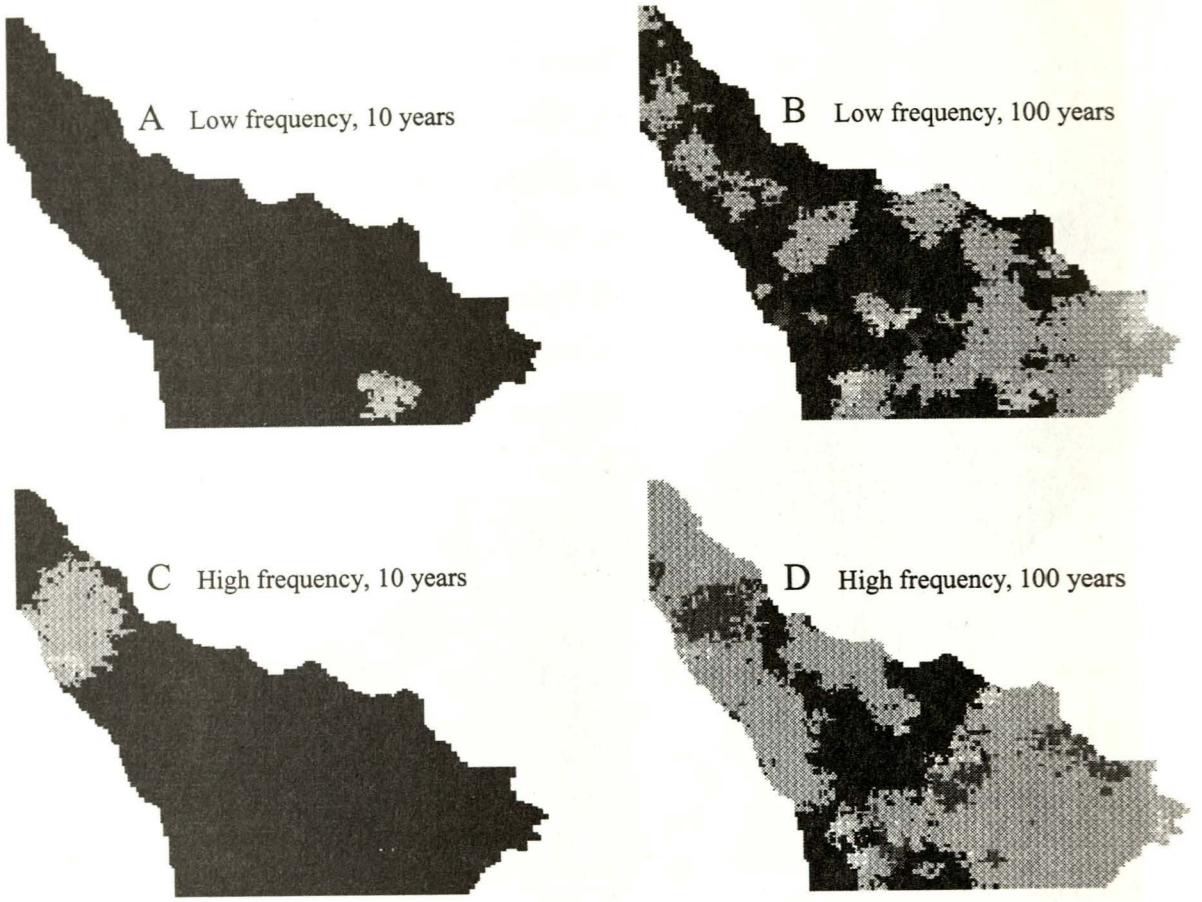


Figure 3. One hundred years of simulated fire disturbance on MOFEP Compartments 7 and 8. Lighter shades indicate disturbance. Accumulated burned area per century is 40% in panel B and 70 percent in panel D

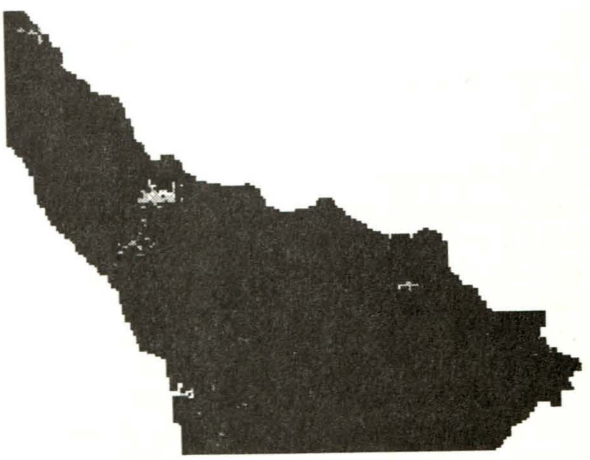

Figure 4. Cumulative simulated wind disturbance over a century. Lighter shades indicate wind disturbance.

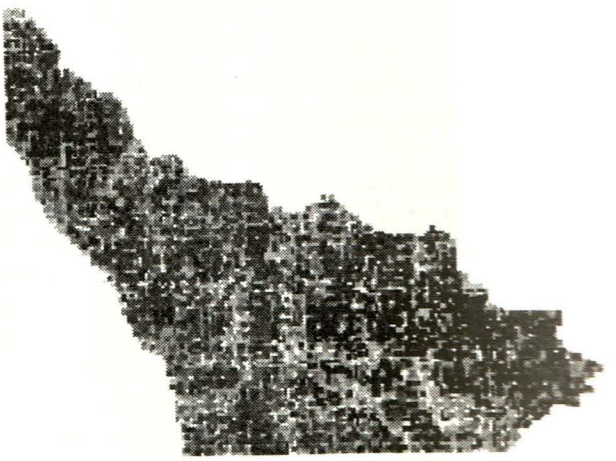

Figure 5. Snapshot in time of simulated age distribution for LANDIS applied to MOFEP Compartments 7 and 8. Darker sites have young cohorts; lighter shades indicate older cohorts.

While Figures 3 through 7 illustrate some of the capabilities and limitations of the modeling approach, model validation is a necessary step prior to large-scale applications. Long-term, spatially explicit databases that would allow direct comparison of model predictions with field observations simply do not exist. However, several sources of information can be used to evaluate model performance. These include minimally disturbed old-growth forests, a few managed forests with known disturbance histories spanning 5 decades, and opinions of foresters and ecologists with extensive experience in the region. Validation and fine-tuning of coefficients is ongoing.

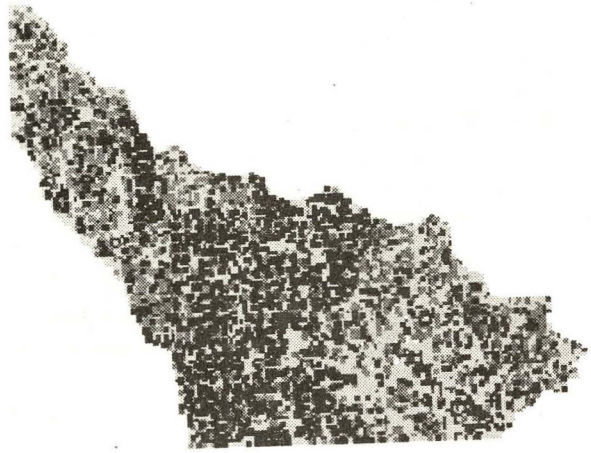

Figure 6. Snapshot in time of dominant species for LANDIS simulation. Shades of gray indicate differing dominant species cohorts

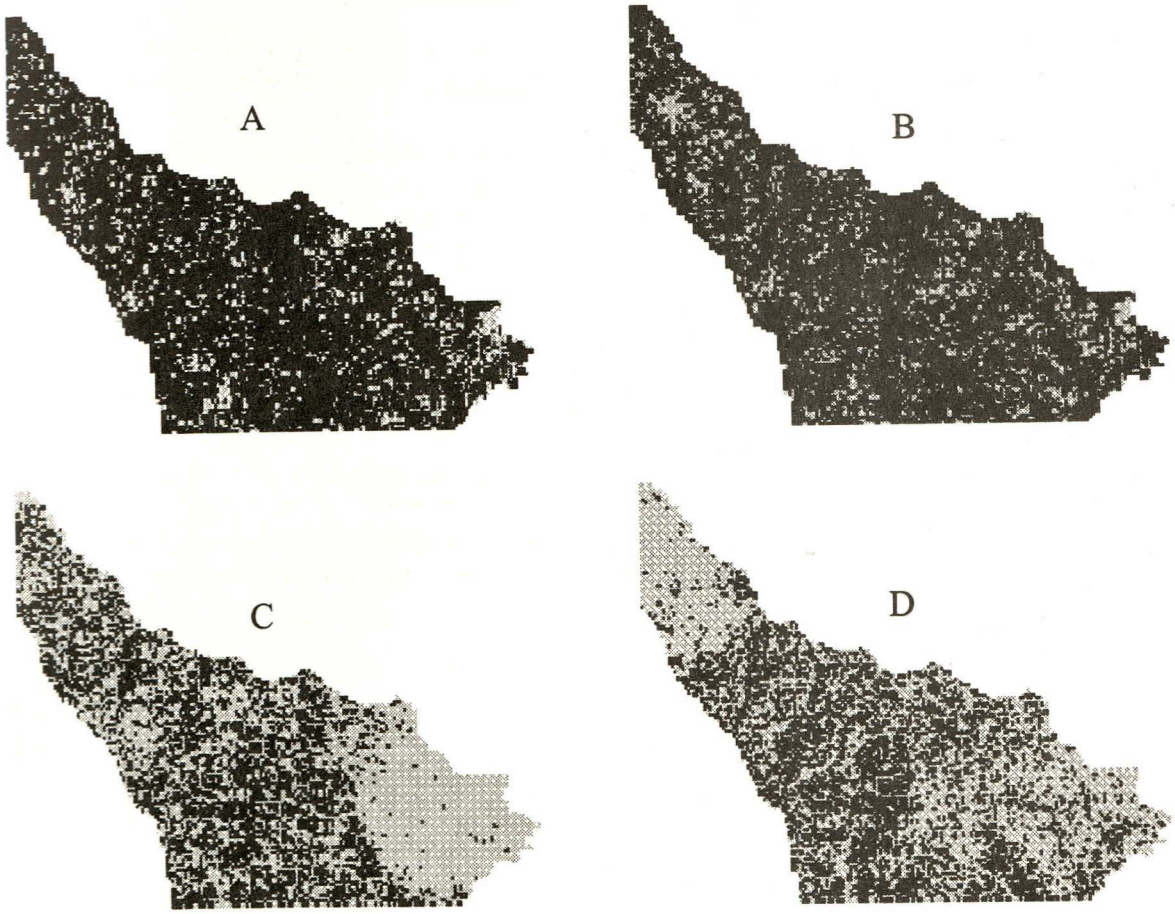


Figure 7. Presence of white oak and red maple species cohorts by site (pixel). Panels A and B show presence (dark sites) of white oak at two times separated by 50 years of simulated change. Panels C and D show presence (dark sites) of red maple for the same sites and time periods. Simulated fires have reduced the presence of red maple in the lower right corner of panel C and the upper left corner of panel D while showing little impact on white oak in panels A and B.

## CONCLUSIONS

We believe it is important for research in support of ecosystem management in the Central Hardwood region to (1) synthesize the rich body of knowledge relevant to Central Hardwood ecosystems; (2) include syntheses that cross the functional lines that have often separated foresters, ecologists, economists, sociologists, and soil scientists; (3) incorporate a modeling framework to organize information; and (4) focus on the big (landscape) scale before getting buried in the site-specific details. The complexity of this research requires a team of cooperators.

Disturbance by fires, harvest, and wind is the norm, not the exception, in Central Hardwood landscapes. Vegetation response to disturbance is at least partially predictable, and many other ecosystem attributes of interest at the stand and landscape scale can be predicted from knowledge of the vegetation composition and structure. Consequently, models that focus on simulating spatially explicit response to disturbance can be useful tools for dealing with the issues inherent in ecosystem management. We anticipate that for maximum utility, such models must be capable of dealing with landscapes as large as 100,000 ha.

Our initial parameterization of the LANDIS simulation model for conditions found in the Missouri Ozarks illustrates the type and format of information contained in landscape models. The technique provides a mechanism for modeling vegetation change through time and predicting response to fire and wind disturbance. The model must be modified to simulate harvest disturbance, but we believe that tools of this kind have the capability to give us a better long-term, large-scale picture of the impacts of disturbance on forest landscapes.

## ACKNOWLEDGMENTS

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# MICROPROPAGATION OF JUVENILE AND MATURE AMERICAN ELMS FROM STEM NODAL SECTIONS

Ann M. Chanon, Joseph C. Kamalay, and Pablo Jourdan[1]

Abstract: A micropropagation system has been established for the regeneration of superior trees of *Ulmus americana* L., the American elm. The development of a reliable regeneration system is also required as a prelude to genetic transformation of selected genotypes. The effects of explant source, media formulations, and plant growth regulator concentrations on shoot regeneration of juvenile and mature American elm selections were examined. Hypocotyls, leaves, and stems taken from seedlings and stem segments taken from root cuttings of a mature tree, selection #8630, were used. Media formulations included Murashige and Skoog (MS) medium, Driver and Kuniyuki Walnut (DKW) Medium, and McCown's Woody Plant Medium (WPM). Growth regulators included benzyladenine (BA) and 1-phenyl-3-(1,2,3-thiadiazol-5-yl) urea (thidiazuron or TDZ). Hypocotyls and nodal stem sections were cut and cultured on MS supplemented with either 2.2 or 4.4 µM BA. Leaves were wounded and cultured on either MS, half strength MS, DKW, or WPM containing TDZ at concentrations ranging from 0.01 to 22.5 µM. Nodal stem sections from the shoots of seedlings and mature trees and regenerated shoots within 4 weeks on both BA concentrations. The BA concentration used had a significant effect on the response of the explants. The number of stem explants proliferating shoots as well as the number of new shoots generated was larger on 2.2 µM BA than on 4.4µM BA. The 4.4µM BA concentration decreased both the quantity and quality of new shoots regenerated. Shoots were readily rooted ex vitro in soilless medium with a Rootone dip. We conclude that micropropagation and tree regeneration can be achieved from both mature American elms and from juvenile seedlings.

## INTRODUCTION

*Ulmus americana* L., the American elm, was a common tree in the urban landscape and hardwood forest across its native North American range until the introduction of Dutch elm disease (DED). DED, caused by *Ophiostoma ulmi* (Buisman), has killed millions of trees. Since American elm seedlings have a pseudo resistance to DED in the juvenile phase, DED tolerance can only be reliably tested on mature trees. Townsend and others (1995) have recently released two DED tolerant American elms, 'New Harmony' and 'Valley Forge'. These elite trees must now be propagated for the nursery industry. A wide array of traditional methods can be used to propagate selected clones, but these techniques are difficult and often unreliable for mature trees of most hardwood species, including the American elm. Poor adventitious rooting and/or slow shoot growth are major limitations in regenerating propagules from older trees (Brand and Lineberger 1992).

A clonal propagation method with American elms is also essential for targeted DNA transformations using either *Agrobacterium*-mediated or direct DNA transfer. DNA transformation is an important approach for elm improvement because of the genetic and reproductive barriers that prevent the creation of disease-resistant trees by conventional plant breeding methods (Ager and Guries 1982). In this case, juvenility or maturity would not be a consideration, but the technical ease of successful propagation would be essential due to the relatively low fraction of genetically transformed cells produced. Until now, the only successful regeneration protocols for American elm have utilized seedling tissue with hypocotyl tissue having the highest regeneration potential. Shoot regeneration was reported from hypocotyl-derived cell suspension cultures (Durzan and Lopushanski 1975); from hypocotyl callus cultures (Karnosky and others 1982); and from cryogenically preserved hypocotyl callus (Ulrich and others 1984).

[1] The authors are Graduate Research Assistant, The Department of Horticulture and Crop Science, The Ohio State University, Columbus, OH 43210; Research Biologist, USDA Forest Service, NEFES, Delaware, OH 43015; and Associate Professor, The Department of Horticulture and Crop Science, The Ohio State University, Columbus, OH 43210.

Ho (1985) produced shoots from seedlings using epicotyl derived callus, but only a small number of shoots were recovered. George and Tripepi (1994) found that leaves had the potential to regenerate shoots, but the efficiency of regeneration was variable.

Micropropagation has been utilized to propagate a diverse group of mature woody plants including *Betula* (Brand and Lineberger 1992), *Larix x eurolepis* (Brassard and others 1996), *Malus* (Abbot and Whitely 1976), *Vitis viniferia* (Mullins and others 1979), and *Liquidambar styraciflua* (Sutter and Barker 1983). Micropropagation and shoot regeneration from tissues of mature trees have been reported for several other elm species but not for *U. americana*. Shoot multiplication from axillary buds was reported for *U. campestris* (Biondi and others 1984) and for *U. glabra* (Chalupa 1979). Lin and others (1981) were able to regenerate plantlets from a 10 year old specimen of *U. japonica*. In 1982, Chalupa demonstrated the use of root suckers as a source of explant material for the propagation of mature *U. campestri*. Proliferating shoot cultures were established from shoot tips of mature *U. procera* (Fenning and others 1993). Both Corchete and others (1993) and Cheng and Shi (1995) used nodal explants of a mature *U. pumila* to initiate shoot proliferating cultures.

The goal of this research was to develop a rapid and efficient micropropagation protocol for both seedling and mature genotypes of American elm. The data presented in this report, which include explant initiation efficiency, the mean number of shoots per primary explant, subsequent subculture regeneration, and rooting ability, indicate the feasibility of micropropagation of American elm.

## MATERIALS AND METHODS

### Plant Material And Explant Preparation

Cleaned and dewinged *U. americana* seed was obtained from Schumacher Seed Co., Sandwich, MA. and was stored dry in the dark at -20° C to maintain viability until needed. According to the supplier, the seed was collected from a number of trees, mixed before processing, and reported to have a 77% germination rate. Groups of 100 seeds were wrapped in bridal netting and surface sterilized for five minutes in a solution of 1.31% sodium hypochlorite and 0.5% Alconox detergent (Alconox, Inc., NY), followed by three rinses in sterile, distilled water. The seeds were germinated on slightly moistened sterile filter paper in plastic petri plates. Germination plates were sealed with plastic wrap and incubated in the dark. After 5 days, the seeds with protruding radicals were transferred to small glass jars containing 25 ml of the basal medium (described later) to complete germination.

For mature American elm propagation, root cuttings were taken from selection #8630 located at the Forest Science laboratory in Delaware, OH. This 36 year old specimen tree was first selected for elm yellows resistance (Schreiber, personal communication) and is in trials to determine DED tolerance levels. The root sections were taken in mid-November and prepared for cuttings. Root pieces were rinsed in tap water to remove soil and debris and were cut into sections 8 to 10 cm in length. Paying careful attention to polarity, the distal ends of the cuttings were dipped into a rooting hormone containing 0.3% indole-3-butyric acid (IBA), were placed in soilless media, and were maintained under culture room conditions (Schreiber 1963). Actively elongating shoot tips (2-6 cm in length) were harvested from the root cuttings, the leaf blades were removed, but the petioles were retained. The shoots were washed with a 2% Alconox solution and surface sterilized by gentle shaking in a 1.05% sodium hypochlorite and 0.5% Alconox solution for 15 minutes, then rinsed three times in sterile distilled water. Nodal segments were prepared from the shoots by trimming any damaged tissue, the petioles, and the shoot apex.

### Culture Media

The basal medium consisted of Murashige and Skoog (MS) salts, (Murashige and Skoog 1962), 100 mg/l myo-inositol, 3% (w/v) sucrose, and 0.75% (w/v) Difco Bacto agar. The shoot initiation medium was basal medium supplemented with either 2.2 µM or 4.4 µM benzyladenine (BA). Additional media were used for shoot regeneration from leaves including: half strength MS salts, Driver and Kuniyuki Walnut (DKW) Medium (Driver and Kuniyuki 1984), and Woody Plant Medium (WPM) (Lloyd and McCown 1980). These were supplemented with myo-inositol, sucrose, and agar as outlined above. The plant growth regulator treatments consisted of thidiazuron

(TDZ) at 0.0, 0.01, 0.1, 1.0, 5.0, 10.0, 15.0, and 22.5 μM. Three different media were used for rooting: basal MS solidified with 0.8% (w/v) agar; MS with 1g/l activated charcoal; and MS with 2.5 μM IBA. Prior to autoclaving, the pH for the MS media was adjusted to 5.7, 5.5 for the DKW, and 5.3 for the WPM.

Shoot Proliferation

Cultures were maintained at 25° ± 2° C under cool white fluorescent lights with a 16 hour photoperiod. The regeneration capacity of 4 week old seedling leaves, hypocotyls, and nodal stem segments was tested, while the roots and cotyledons were discarded. In a preliminary study, the hypocotyl and nodal stem segments were placed on shoot initiation medium with either 2.2 or 4.4 μM BA at a density of one explant per glass jar containing 25 ml of medium. In subsequent seedling studies, 8 to 10 hypocotyl or stem segments were used per GA7 vessel (Magenta Corp., Chicago, IL) containing 50 ml of medium with 25 or 30 replicates per treatment.

Nodal segments from #8630 were placed horizontally at a density of two to four segments per GA7 vessel on MS with either 2.2 or 4.4 μM BA for shoot proliferation. The newly formed shoots were routinely subcultured on MS with 2.2 μM BA at a density of nine explants per GA7 vessel every 4 weeks for further shoot proliferation.

Using the George and Tripepi (1994) protocol, seedling leaves were taken from the fourth, fifth, sixth and seventh nodes and were wounded by cutting across the mid-vein and piercing the lamina. Leaves were cultured on one of the four different basal media with one of the seven different TDZ concentrations or a control without TDZ for a total of 32 treatment combinations. The leaves were plated abaxial side up at a density of four leaves per plate with 15 replicate plates per treatment. The plates were placed in complete darkness for 14 days and then moved to the culture shelf conditions described above. Explants were examined for 8 weeks for shoot regeneration.

Rooting And Plantlet Production

Healthy shoots approximately 4 cm in length with fully expanded leaves from both the seedling and #8630 cultures were placed in one of five rooting treatments. Rooting treatments consisted of three in vitro treatments on modified MS media described above and two ex vitro treatments. For the in vitro treatments, nine microcuttings per replicate per treatment were aseptically placed vertically in approximately one centimeter of media in a GA7 vessel. For the ex vitro treatments, microcuttings were washed under running water to remove any adhering medium. The basal end of the cutting was dipped in water and either placed into the Redi-Earth or dipped into Rootone (Rigo Co., Buckner, KY) and then placed into the Redi-Earth. The microcuttings were stuck in 3 inch by 5 inch aluminum trays with plastic lids with 25 microcuttings per tray.

The microcuttings were rooted under culture room conditions. After 4 weeks, the microcuttings were inspected and rooting percentages recorded. Acclimation of rooted microcuttings was completed by repotting the plantlets into cell packs containing soilless medium and placing them under intermittent mist (6 seconds mist every 6 minutes) and 60% light exclusion for 10 days followed by 14 days under 60% light exclusion without mist. The acclimated plants were maintained in the greenhouse under full sun following standard horticultural practices.

Data Analysis

Data were taken after 4 weeks on the number of explants that initiated shoot regeneration and the number of shoots per regenerating explant for both the seedling and #8630 experiments. The data were analyzed using the SAS General Linear Model (GLM) procedure (SAS Institute 1992) to assess treatment differences and shoot regeneration responses. The means and percentages were separated using both T-tests (LSD) and Duncan's multiple Range Test.

## RESULTS

### Shoot Proliferation From Seedling Explants

The differences in shoot proliferation response of the three explant sources were quite apparent. The preliminary experiment indicated that shoot proliferation, which could have resulted from axillary, adventitious, or both sources, was initiated from stem nodal segments but not from hypocotyl tissue. Further, explants on the lower BA concentration produced an average of 4.5 shoots per responding explant while those on the higher BA concentration produced 1.8 shoots per responding explant (data not shown). In the second experiment, both BA levels tested induced shoot proliferation on the stem explants (Table 1), but 2.2 μM BA was significantly better than 4.4 μM BA in both the percentage of responding explants and the mean number of new shoots per responding explant. Of the explants tested, 78.5% responded on the medium containing 2.2 μM BA while 39.0% responded to the medium containing 4.4 μM BA. Nodal segments exposed to 2.2 μM BA produced on average 2.3 shoots, while 1.7 shoots were produced on medium with 4.4 μM BA

Table 1. The effect of seedling explant source and BA concentration on shoot initiation and multiplication

| Explant source | Culture medium | No. responding/ no. cultured | Percentage of explants responding | Mean no. of new shoots /responding explant ± SE |
|---|---|---|---|---|
| Hypocotyl | MS 2.2 μM BA | 1/270 | 0.4 | 0 |
| | MS 4.4 μM BA | 0/252 | 0.0 | 0 |
| Stem | MS 2.2 μM BA | 204/258 | 78.5*** | 2.3 ± 0.6** |
| | MS 4.4 μM BA | 71/184 | 39.0 | 1.7 ± 0.7 |

*** Percentages are significantly different at $\alpha = 0.001$.
** Mean numbers are significantly different at $\alpha = 0.01$

The results of the preliminary experiment indicated that the hypocotyl tissue responded poorly to the conditions used. When the experiment was repeated with a larger number of explants, the trends remained consistent. Only one out of the 540 hypocotyls tested produced any callus and it failed to undergo shoot organogenesis. The other hypocotyls darkened and senesced without further development.

In an effort to replicate the results of George and Tripepi (1994), seedling leaves were used as an explant source. Wounded leaves began to proliferate callus within 1 week of plating on all TDZ concentrations tested. During the 2 week dark period, friable and creamy white callus formed on the wound sites, along the midrib, and at the base of the petiole. After exposure to light, the callus began to darken and by the fifth week was dark brown and no growth was apparent. A second type of callus was sporadically observed on the explants plated on 10 and 15 μM TDZ concentrations. This callus was hard, green, and grew rapidly in the light where it quickly covered the softer callus. The various media formulations did not appear to affect callus formation. This protocol was repeated four separate times but unlike George and Tripepi, no shoot organogenesis was observed from the leaf tissue itself or from either type of callus.

### Shoot Proliferation From #8630

While shoot proliferation was observed from #8630 explants cultured on both BA concentrations, the 2.2 μM BA was significantly better than 4.4 μM BA in both the percentage of responding explants and the mean number of new shoots per responding explant (Table 2). An overall average number of 3.9 shoots were produced per explant on 2.2

μM BA, while only 0.6 shoots were produced per explant on 4.4 μM BA. The shoots produced on medium containing 4.4 μM BA were smaller with shorter internodes and decreased leaf area.

The response of the nodal sections cultured from the initial cambium sprouts of #8630 root cuttings differed from the subsequent cuttings from the same root segments. While the percentage of explants proliferating shoots was not significantly different, the number of new shoots per explants was highly significant. Explants harvested 1/25/96 produced a range of 3-16 new shoots per responding explants on 2.2 μM BA while explants placed on 4.4 μM BA produced callus but did not proliferate shoots. Explants from the other harvest dates produced fewer shoots. The responses of the subcultured nodal sections of #8630 were similar to the those of the seedlings.

Table 2. The effect of BA concentration on shoot initiation and multiplication of #8630 explants

| Culture medium | Date of initiation | Total no. of explants | Percentage of explants responding | Mean no. of new shoots/responding explant ± SE |
|---|---|---|---|---|
| MS 2.2 μM BA | 1/25/96 | 17 | 47.6 a[1] | 9.1 ± 4.1 b |
| | 2/14/96 | 58 | 61.1 a | 2.9 ± 2.0 c |
| | 3/19/96 | 17 | 55.5 a | 1.9 ± 1.7 c |
| | 4/03/96 | 18 | 55.6 a | 2.5 ± 1.5 c |
| Overall | | 110 | 55.7 *[2] | 3.9 ± 3.6 **[3] |
| MS 4.4 μM BA | 1/25/96 | 12 | 8.3 | 0.0 |
| | 2/14/96 | 28 | 50.0 | 0.8 ± 0.7 |
| Overall | | 40 | 28.1 | 0.6 ± 0.7 |

[1] Means separation within a column followed by the same letter are not significantly different using Duncan's multiple range test at P = 0.05.
[2] Overall percentages are significantly different at $\alpha$= 0.05.
[3] Overall mean numbers are significantly different at $\alpha$ = 0.01.

Repeated Subculture

The results of subculturing from seedling-derived and #8630-derived shoots are presented in Figure 1. The results were similar for both seedlings and #8630 shoots. An average of 3.0 new shoots was observed for every seedling nodal explant used, while an average of 3.6 new shoots was observed for every #8630 nodal explant used. There were no significant changes in the extent or quality of shoot production during continued subculture of the #8630-derived shoots.

For the seedling population, shoot proliferation seemed to improve with repeated subculture. The second passage of subculturing produced an average of 4.1 shoots per responding explant compared to an initial average of 2.3 shoots per responding explant. This may indicate a selection for plants that will grow better in culture. For the #8630, shoot proliferation did not appear to be affected by repeated subculture. An average of 3.5 new shoots developed for every explant used and this was consistently observed.

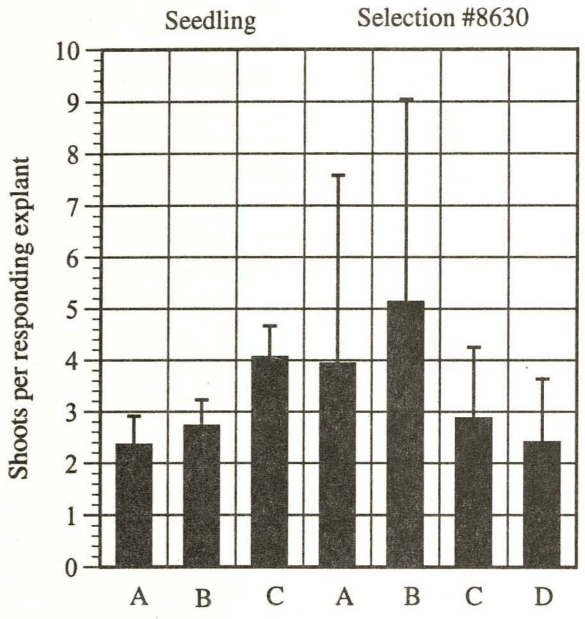


Figure 1. Comparison of shoot multiplication rates between the seedling population and #8630. Histograms indicate the mean number of shoots from initiation and subsequent subcultures on MS supplemented with 2.2 µM BA. Vertical lines indicate ± SE; **A:** initiation; **B:** first subculture; **C:** second subculture; **D:** third subculture.

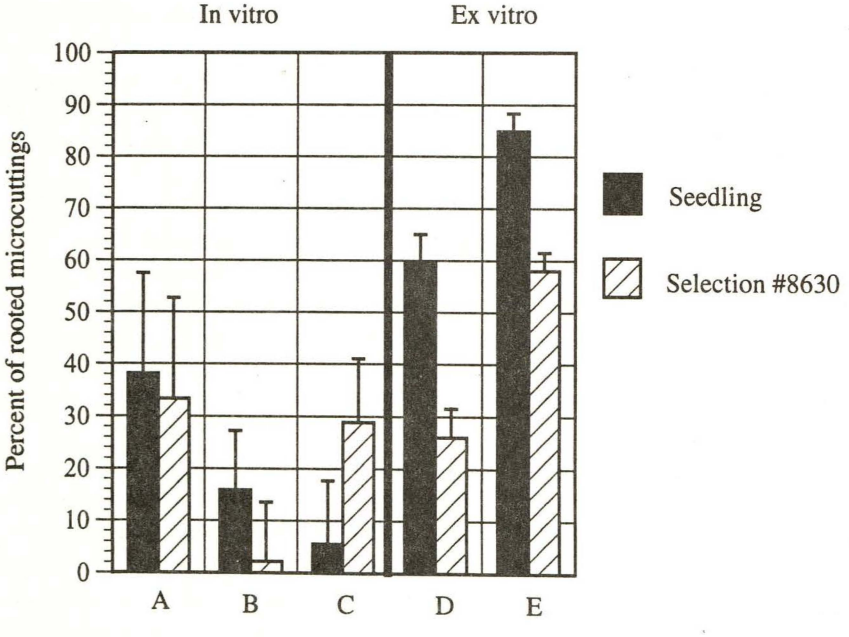


Figure 2. Comparison of microcuttings rooted from the seedling population and #8630. Histograms indicate the mean percentage of rooted microcuttings observed within 4 weeks of treatment. Vertical lines indicate ± SE; **A:** MS with 0.8% Agar; **B:** MS with 1 g/L activated charcoal; **C:** MS with 2.5 µM IBA; **D:** Redi-Earth without Rootone; **E:** Redi-Earth with Rootone.

Rooting And Plantlet Production

As illustrated in Figure 2, the most effective treatment for rooting American elm microcuttings was Redi-Earth with a Rootone dip. Within 2 weeks, leaf abscission and shoot death were observed on microcuttings of both #8630 and the seedling population placed on in vitro rooting medium with activated charcoal. A similar response was observed for the seedling microcuttings on medium containing 2.5 µM IBA. Rooting percentages were affected by the rooting treatment. For the seedling microcuttings, all in vitro treatments rooted at less than 40% while the ex vitro treatments showed significantly higher rooting percentages. Eighty-five percent of the seedling-derived microcuttings rooted when stuck in Redi-Earth after a Rootone dip while 59% rooted without Rootone treatment. The trends for the #8630-derived shoots were similar, but the percentages of rooted microcuttings were generally lower than the seedling-derived microcuttings.

## DISCUSSION

Nodal stem tissue from both juvenile and mature American elm selections were adequate starting materials for micropropagation. Axillary shoots have been shown to be the most reliable explant source to produce clonal plants from tissue culture (Brand and Lineberger 1992). Chalupa (1982) also demonstrated that stump and root sprouts were good sources of explant material and had a higher regeneration potential than explants taken from other parts of a mature tree, *U. glabra*. Previous work with above ground tissues from #8630 have consistently failed to produce propagules despite various techniques including softwood cuttings, hardwood cuttings, and grafts. Root cuttings and the micropropagation of shoots originally derived from root cuttings have resulted in the clonal propagation of this American elm.

In the current study, shoot regeneration was achieved on MS medium supplemented with either 2.2 µM or 4.4 µM BA, but explants on the lower concentration of BA produced longer shoots with larger leaves. Both Corchete and others (1993) and Cheng and Shi (1995) found that lower concentrations of BA stimulated shoot formation on explants of *U. pumila* better than higher concentrations of BA in their respective studies. No obvious mutants were observed from the shoots regenerated in vitro and all shoots were approximately the same size prior to harvest as microcuttings. As a group, the ex vitro treatments rooted significantly better than the in vitro treatments. Even those in vitro treatments which resulted in similar levels of rooted cuttings were less desirable than the ex vitro treatments due to the relatively low cost and ease of the latter method. The best rooting was achieved using a soilless mix and a Rootone dip.

Only one of the three seedling tissues utilized, showed repeatable shoot proliferation. The hypocotyl explants failed to form callus or undergo shoot organogenesis on MS with either BA concentration. This result differs from Durzan and Lopushanski (1975) and Ulrich and others (1984), who observed both callus production and shoot regeneration from seedling hypocotyl tissue. Differences in growth may be the result of genotype effects, differences in seedling age, or the differences in plant growth regulator concentrations, particularly their use of an auxin during callus initiation.

Seedling leaf explants failed to support shoot proliferation in culture. George and Tripepi (1994) reported significant levels of shoot initiation from seedling leaf explants under similar conditions using leaves from 2 year old half-sib seedlings from an open-pollinated Delaware-2. They reported that some leaf explants supported shoot proliferation on DKW supplemented with 7.5, 15.0, or 22.5 µM TDZ, but not all genotypes responded. Both the age and the genotype of George and Tripepi's explants differed from the seedling used in this study. Despite the failure of shoot proliferation from leaf tissues in these experiments, different methods of shoot regeneration from leaf tissue are being examined as a preliminary step to plant transformation.

Note: RAPD DNA profiles (Kamalay and Carey. 1995. J. Environ. Hort. 13: 155-159) of the cuttings propagated from the #8630 roots do not match those taken from the leaves of the mature tree. At the time of this writing, this discrepancy cannot be explained but further research is being conducted to answer this question.

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# ESTABLISHMENT OF NORTHERN RED OAK GENETIC TESTS WITH NURSERY-GRADED SEEDLINGS

S.A. Lay[1], M.A. Remaley[1], S.E. Schlarbaum[1], P.P. Kormanik[3], T. Tibbs[4], R.A. Cox[5], T. LaFarge[4], and A.M. Saxton[2]

Abstract: Artificial regeneration of northern red oak (*Quercus rubra* L.) has had variable success over time. Current nursery practices generally involve the growth of seedlings to a standardized height and form with little regard to seed source, seedling quality, or subsequent field performance. Additionally, there is not an accepted culling criteria for hardwood nursery stock. Northern red oak seedlings need sufficient height and root growth in the nursery in order to receive adequate sunlight, water, and nutrients when planted in the field. In the Fall of 1993, acorns were collected from 73 individual trees in a seedling seed orchard on the Watauga Ranger District of the Cherokee National Forest. The acorns were planted by family at the Georgia Forestry Commission's Flint River Nursery, and grown according to prescriptions developed by the USDA Forest Service Institute for Tree Root Biology. The seedlings were lifted after one growing season and visually graded according to number of first order lateral roots (FOLR), tap root size, basal caliper, and stem height. Lateral roots were pruned 6 inches from the tap root to minimize J-rooting. In the late winter of 1995, plantations were established on southern National Forests (16), a Kentucky State Forest (1), and the Ames Plantation in West Tennessee (1). At each location, 25 families were represented with a total of 1,250 high quality seedlings that were planted using a shovel or auger. Eleven of the 18 plantations are still viable after two growing seasons. The effectiveness of artificial regeneration of quality grown graded northern red oak nursery stock will be evaluated. Differences in outplanting performance among provenances and genetic families will be determined.

[1] Department of Forestry, Wildlife and Fisheries, Tennessee Agricultural Experiment Station, The University of Tennessee, Knoxville, TN 37901-1071.

[2] Statistical and Computing Services, Tennessee Agricultural Experiment Station, The University of Tennessee, Knoxville, TN 37901-1071.

[3] USDA Forest Service, Southern Research Station, Institute of Tree Root Biology, Forestry Sciences Laboratory, Carlton Street, Athens, GA 30602.

[4] USDA Forest Service, Southern Region, 1720 Peachtree Rd., Atlanta, GA 30367.

[5] Tennessee Department of Agriculture, Division of Forestry, P.O. Box 2666, Knoxville, TN 37901-2666.

# INFLUENCE OF WEATHER ON POLLINATION AND ACORN PRODUCTION IN TWO SPECIES OF MISSOURI OAKS

Robert A. Cecich[1]

Abstract: The process by which oak pistillate flowers become acorns is reasonably understood from an anatomical perspective; however, the way that various factors influence this process is still unclear. This study examined acorn production in a small population of white oak and black oak trees in central Missouri, from 1990 to 1995, in relation to weather variables (maximum and average temperature, relative humidity, fog, rain, and hail) at the time of pollination. There was large tree-tree and year-year variation in the size of flower crops and the dates when the flowers aborted. In the white oaks, acorn production was never high in relation to the number of flowers produced; most flowers aborted by July every year. The black oaks generally produced more acorns than the white oaks, but some individual trees never produced a mature acorn from their large flower crops. This variation in acorn production is correlated with certain weather variables at the estimated time of pollination and for the 1 or 2 weeks before and after those dates. Significant negative correlations with black and white oak acorn production were found for maximum temperature and the number of days with hail during pollination. Rain during pollination was negatively correlated with white oak flower survival in July. In both the black and white oaks, flower survival in early July, following pollination, was positively correlated with acorn production.

## INTRODUCTION

Reliably predicting an acorn crop more than a few weeks before the acorns drop from a tree has proven to be a difficult task (Cecich 1993, Gysel 1958, Koenig and others 1994b). Feret and others (1982) concluded that production of white oak acorns could best be predicted from the number of peduncles borne per shoot. Sork and Bramble (1993) and Sork and others (1993) studied flower and acorn production in the same populations of northern red (*Q. rubra* L.), black (*Q. velutina* Lam.), and white (*Q. alba* L.) oaks for 8 years and provided evidence that these three species have inherent cycles of reproduction that are affected by weather and previous reproductive history—black oak with a 2-year cycle, white oak with 3-year cycles, and northern red oak with 4-year cycles. Koenig and others (1994a) have provided similar results for oaks in the Western United States. Although numerous pistillate flowers in the spring do not guarantee a large acorn crop (Sharp 1958, Sharp and Chisman 1961, Wright 1953, Gysel 1956, Cecich and others 1991), few-to-no flowers means a small to nonexistent acorn crop (Sork and Bramble 1993).

A necessary step to developing confidence in predicting an acorn crop is an awareness of major yield components of the acorn production process. These include: (1) *flower initiation*, the process whereby chemical, genetic, and abiotic factors interact during a critical time period to cause a meristem to commit itself to become a flower or flower part (Mandel and Yanofsky 1995, Weigel and Nilsson 1995); (2) *differentiation*, the structural manifestation of the initiation process; e.g., the development of the staminate and pistillate inflorescences and their flower primordia (Merkle and others 1980); (3) *emergence* of the flowers in spring (the first visible phase of the flowering process) associated with receptivity of the stigmas and shedding of pollen; (4) *fertilization* of the egg by the male gamete; and (5) *maturation*, the development of the flower and embryo into an acorn. In central Missouri, pollination occurs in late April to early May. Fertilization occurs in mid-June of the first growing season in white oak (Lepidobalanus subgenus) and in late June of the second growing season in black oak (Erythrobalanus subgenus) (Cecich, in press). Mortality occurs at every stage of development, but up to 100% of flower loss can occur by the time of fertilization (Williamson 1966, Kossuth 1974, Cecich 1991).

[1] Research Plant Physiologist, USDA Forest Service, North Central Forest Experiment Station, Columbia, MO 65211.

Weather affects acorn production. Emergence of staminate inflorescences and shedding of pollen increase or hasten with rising temperatures and the associated lower relative humidity (Romashov 1957, Sharp and Chisman 1961, Jovanovic and Tucovic 1975). Relative humidity (rh) can limit the size of an acorn crop. When Wolgast and Stout (1977) controlled rh at the time of pollen shed and stigma receptivity in a growth chamber, no acorns matured when rh exceeded 61%, but about half the flowers matured into acorns when rh was lower. Sharp and Sprague (1967) found no correlation with rh and acorn yields in field studies and concluded that temperature was a primary factor in determining the success of acorn crops. Temperature may directly or indirectly influence the time of flower emergence through branch and leaf elongation (Minina 1954, Cecich 1993). Sharp (1958) found that low temperatures in the spring did not affect flowering unless there was a freeze sufficient to damage shoots and leaves. Late-spring freezes can kill first-year (Sork and Bramble 1993) or second-year (Wolgast and Trout 1979) pistillate flowers in the Erythrobalanus subgenus. Sharp and Sprague (1967) considered the impact of drought on differentiation of flowers and developing acorns to be inconclusive; however, Sork and others (1993) found a negative correlation of summer drought, a combination of temperature and rainfall effects, with acorn production.

How can we predict the size of an acorn crop more efficiently? Since most flower abortion occurs between pollination and fertilization, what weather factors are most likely to affect the pollination process? How might these factors and their timing or duration be related to the development of the flower? Any such speculation should be accompanied by some notation of time; i.e., during what specific time period is that factor operational? This study examines the impacts of several weather factors (maximum and average temperature, relative humidity, rain, and hail) 1 and 2 weeks before, during, and 1 week after the time of pollination, on the survival of flowers into acorns.

## METHODS

Study trees (12 black oaks and 9 white oaks), from 10 to 25 m tall, were located along a forest road on the Thomas S. Baskett Wildlife Research and Education Area, Ashland, Missouri (38°N, 92°W). Sampling of the oak flowers was done from a bucket lift on a truck equipped with a 13-m hydraulic boom that reached to a maximum height of 16 m. Flower populations from 1990 to 1995 and their survival were monitored weekly from pollen shed to acorn maturation. On each tree, 5 current-year branches on a major branch system were selected for monitoring flower survival. The total flower survival of the five branches was recorded each week. For this study, 1994 was the last flower crop for black oak (acorns matured in 1995) and 1995 was the last year for white oak.

The dates of pollen shed for the study trees varied by species, from tree to tree within a species, and by year. Since controlled pollinations were not used to evaluate the timing of developmental events, an approximate date of pollen shed was recorded each year. Each species at the experimental site shed pollen for about 1 week, with peaks of activity for black oak several days earlier than that for white oak.

Climatological data for the months of April and May during the years 1988 to 1995, recorded at the nearby Columbia Regional Airport, were evaluated. Daily weather variables were average temperature (Ave C), maximum temp. (Max C), relative humidity (Hum), days with fog (Fog), days with rain (Rain), and days with hail (Hail). Only 1 night dropped to a low of 0° C, so the minimum temperature data were not used. Since the 2-month period was still too broad a timeframe for evaluating the effects of weather on the pollination process, the weather data for a given year were separated into weekly increments defined by the Estimated Pollination Date (EPD) for that year. For example, if pollen were shed on May 1, the 3 days before and after May 1 would make up the week of the EPD. One and 2 weeks earlier are noted as EPD-1 and EPD-2, respectively; the week after the EPD is noted as EPD+1. The EPD's for black and white oaks were generally about 2-3 days apart, but are shown as the same dates each year for convenience, except for 1993, when black oak shed pollen about 1 week ahead of white oak. Since the weather data were daily averages and the hourly variation could not be assessed, the data were pooled into weekly EPD values that were correlated with stages of flower development and acorn production. Statistical evaluation was done with the JMP software package for the Macintosh (SAS 1994).

## RESULTS

Flower and acorn production in the black and white oaks varied by year and among individual trees (Tables 1, 2). In both species, the largest total flower crop was produced in 1990 and was about 2-3 times as large as subsequent flower crops. The coefficient of variation (cv) for the flower crops over the period of study was the same for both species, but the cv for acorn production was twice as large for white oak (Tables 1, 2).

Most of the black oak flowers in 1990 were produced on five trees (#4, 5, 10, 11, and 12). In subsequent years, trees 11 and 12 remained the "best" flower producers. Although tree 11 was the best acorn producer, the adjacent tree 12 never produced an acorn crop during this study (nor in the 1996 crop not reported here). A large percentage of tree 12's reproductive potential was lost every July and August of the second growing season during the stages of cotyledon and embryo enlargement, indicating the possible expression of late embryonic lethal alleles (Fisher 1965) carried by tree 12. As a contrast to good flowering, tree 7 was not a good flower-producing tree, except for 1993. In 2 consecutive years it did not have pistillate flowers or pollen catkins. Trees 2 and 6 did not have flowers in either 1991 or 1994. Acorn production, when considered as a percent of the number of flowers, varied between 7 and 15%, with an average of about 10% for all years (Table 3).

The heavy white oak flower crop in 1990 was produced on six of the nine trees (Table 2). Tree 8 came closest to regularly producing a good flower crop, but only produced a relatively good acorn crop in 1995. Acorn production for these white oaks, when considered as a percent of the initial flower crop, averaged about 7% over all years (Table 4).

Flower crops for all the black oaks were pooled by year and correlated with survival at the subsequent developmental stages (Table 5). The general tendency within a year was that all correlations with the flower crops were reduced the farther in time the developmental stage got from the flower stage. When the black oak developmental stages were examined across all flowering years, correlations with the acorn crop were significant for the J1 and S1 stages (Table 6).

White oak flower survival in July (J1) for all trees was negatively correlated with acorn production in the previous year ($r = -0.909$) and positively correlated with acorn production 2 years prior ($r = 0.913$). Percent survival of white oak acorns was highly correlated with the percent flower survival in July ($r = 0.956$), but weakly correlated with initial flower numbers ($r = 0.536$).

The weather variables, assembled as weekly averages, are shown in conjunction with the EPD classes in Table 7. Flower survival values between 1990 and 1995 were correlated with days of hail occurrence during the pollination season (includes all the EPD classes). White oak flower survival in July ($r = -0.867$) and acorn production in September ($r = -0.878$) both had significant negative correlations. For black oak, only survival at the J2 stage (July of the second growing season) showed a strong, but non-significant, negative correlation ($r = -0.856$) with hail during pollination; all other stages were weakly correlated.

The daily averages for relative humidity (rh) were generally above the 60 percent threshold described by Wolgast and Stout (1977). Therefore, no further direct investigation of the relationship of rh to pollen shedding and acorn production could be conducted. If hourly values of rh were available, then the "4 hr-low relative humidity" period or window that Sharp and Chisman (1961) described as necessary for pollen release could also have been evaluated. However, the two EPD's that contain rh values below 60% (EPD-1 and EPD-2 in 1994) were associated with the poorest white oak acorn crop during this study (3.6%).

Table 1. Number of black oak pistillate flowers (F) and acorns (A) on 5 branch tips per tree

| | Year and stage of development | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Tree | 90-F | 91-A | 91-F | 92-A | 92-F | 93-A | 93-F | 94-A | 94-F | 95-A | r [1] | |
| 1 | 50 | 2 | 34 | 0 | 57 | 11 | 64 | 17 | 40 | 4 | 0.898 | * |
| 2 | 79 | 10 | 0 | 0 | 63 | 21 | 42 | 8 | 0 | 0 | 0.810 | |
| 3 | 24 | 0 | 44 | 11 | 53 | 2 | 67 | 14 | 62 | 12 | 0.747 | |
| 4 | 190 | 14 | 19 | 0 | 29 | 1 | 52 | 3 | 66 | 0 | 0.954 | ** |
| 5 | 220 | 0 | 55 | 0 | 53 | 0 | 78 | 1 | 63 | 0 | -0.124 | |
| 6 | 49 | 9 | 0 | 0 | 58 | 6 | 34 | 7 | 0 | 0 | 0.901 | * |
| 7 | 13 | 0 | 0 | 0 | 0 | 0 | 68 | 5 | 36 | 6 | 0.850 | |
| 8 | 42 | 1 | 54 | 12 | 72 | 15 | 50 | 2 | 54 | 5 | 0.865 | |
| 9 | 41 | 3 | 0 | 0 | 32 | 4 | 55 | 4 | 42 | 0 | 0.572 | |
| 10 | 262 | 22 | 39 | 4 | 62 | 24 | 50 | 0 | 49 | 8 | 0.593 | |
| 11 | 233 | 49 | 127 | 25 | 93 | 19 | 96 | 18 | 52 | 1 | 0.985 | ** |
| 12 | 318 | 0 | 70 | 0 | 60 | 0 | 86 | 0 | 68 | 0 | 0.000 | |
| Mean | 126.8 | 9.2 | 36.8 | 4.3 | 52.7 | 8.6 | 61.8 | 6.6 | 44.3 | 3.0 | | |
| sd | 109.2 | 14.4 | 37.7 | 7.9 | 23.5 | 9.0 | 18.3 | 6.4 | 23.1 | 4.0 | | |

r[1] Correlation coefficients for flower and acorn crops between 1990 and 1994.
*, ** indicate P>0 .05 and 0.01, respectively.
Coefficient of variation for population means over 5 year period: F = 56%, A = 42%.

Table 2. Number of white oak pistillate flowers (F) and acorns (A) on 5 branch tips per tree

| | Year and stage of development | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1990 | | 1991 | | 1992 | | 1993 | | 1994 | | 1995 | | |
| Tree | F | A | F | A | F | A | F | A | F | A | F | A | r[1] |
| 1 | 74 | 6 | 43 | 4 | 36 | 0 | 38 | 0 | 101 | 0 | 65 | 6 | 0.141 |
| 2 | 82 | 5 | 60 | 15 | 30 | 4 | 27 | 8 | 95 | 0 | 40 | 8 | -0.366 |
| 3 | 248 | 4 | 16 | 3 | 60 | 1 | 0 | 0 | 50 | 0 | 0 | 0 | 0.614 |
| 4 | 158 | 0 | 65 | 0 | 83 | 1 | 0 | 0 | 46 | 0 | 139 | 11 | 0.466 |
| 6 | 203 | 8 | 37 | 3 | 83 | 5 | 85 | 6 | 65 | 3 | 94 | 16 | 0.350 |
| 7 | 34 | 0 | 86 | 1 | 26 | 0 | 50 | 2 | 47 | 0 | 63 | 9 | 0.373 |
| 8 | 103 | 1 | 56 | 1 | 45 | 0 | 66 | 4 | 107 | 0 | 167 | 32 | 0.814 * |
| 9 | 220 | 6 | 30 | 0 | 30 | 1 | 55 | 1 | 146 | 0 | 39 | 6 | 0.376 |
| 11 | 195 | 62 | 0 | 0 | 37 | 9 | 44 | 13 | 127 | 0 | 61 | 15 | 0.722 |
| Mean | 146.3 | 10.2 | 43.7 | 3.0 | 47.8 | 2.3 | 40.6 | 3.8 | 87.1 | 0.3 | 74.2 | 11.4 | |
| sd | 75.3 | 19.6 | 26.4 | 4.7 | 22.4 | 3.1 | 28.3 | 4.5 | 36.9 | 1.0 | 51.8 | 9.1 | |

r[1] Correlation coefficients for flower and acorn crops from 1990 to 1995.
* = P> 0.05.
Coefficient of variation for population means over 6-year period: F = 55%, A = 87%.

Table 3. Percent survival at various stages of black oak flower development into acorns

| Stage* | Tree No. 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | Mean |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 90-F | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| 90-J1 | 28 | 51 | 38 | 55 | 55 | 31 | 46 | 43 | 56 | 38 | 71 | 43 | 46 |
| 90-S1 | 20 | 39 | 21 | 47 | 46 | 27 | 46 | 40 | 29 | 29 | 61 | 39 | 37 |
| 91-M2 | 20 | 32 | 17 | 45 | 43 | 22 | 38 | 36 | 22 | 26 | 60 | 38 | 33 |
| 91-J2 | 16 | 24 | 13 | 31 | 39 | 22 | 31 | 29 | 22 | 16 | 42 | 33 | 27 |
| 91-A | 4 | 13 | 0 | 7 | 0 | 18 | 0 | 2 | 7 | 8 | 21 | 0 | 7 |
| 91-F | 100 | | 100 | 100 | 100 | | | 100 | | 100 | 100 | 100 | 100 |
| 91-J1 | 44 | | 84 | 63 | 87 | | | 74 | | 46 | 63 | 66 | 66 |
| 91-S1 | 41 | | 80 | 53 | 87 | | | 72 | | 46 | 60 | 64 | 63 |
| 92-M2 | 41 | | 57 | 16 | 82 | | | 72 | | 46 | 56 | 57 | 53 |
| 92-J2 | 12 | | 27 | 5 | 55 | | | 26 | | 31 | 43 | 50 | 31 |
| 92-A | 0 | | 25 | 0 | 0 | | | 22 | | 10 | 20 | 0 | 10 |
| 92-F | 100 | 100 | 100 | 100 | 100 | 100 | | 100 | 100 | 100 | 100 | 100 | 100 |
| 92-J1 | 58 | 84 | 70 | 55 | 77 | 36 | | 75 | 59 | 77 | 78 | 97 | 70 |
| 92-S1 | 53 | 81 | 60 | 52 | 74 | 34 | | 65 | 50 | 73 | 72 | 97 | 65 |
| 93-M2 | 49 | 70 | 55 | 17 | 68 | 26 | | 50 | 28 | 50 | 42 | 58 | 47 |
| 93-J2 | 26 | 56 | 9 | 7 | 51 | 14 | | 33 | 22 | 47 | 32 | 57 | 32 |
| 93-A | 19 | 33 | 4 | 3 | 0 | 10 | | 21 | 13 | 39 | 20 | 0 | 15 |
| 93-F | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| 93-J1 | 56 | 57 | 72 | 27 | 79 | 53 | 37 | 30 | 38 | 48 | 79 | 65 | 53 |
| 93-S1 | 50 | 36 | 57 | 23 | 77 | 35 | 32 | 20 | 29 | 40 | 69 | 59 | 44 |
| 94-M2 | 41 | 24 | 30 | 13 | 60 | 32 | 28 | 6 | 13 | 10 | 54 | 35 | 29 |
| 94-J2 | 34 | 19 | 30 | 10 | 49 | 24 | 12 | 4 | 9 | 8 | 31 | 30 | 22 |
| 94-A | 27 | 19 | 21 | 6 | 1 | 21 | 7 | 4 | 7 | 0 | 19 | 0 | 11 |
| 94-F | 100 | | 100 | 100 | 100 | | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| 94-J1 | 80 | | 71 | 27 | 76 | | 58 | 56 | 48 | 45 | 15 | 40 | 52 |
| 94-S1 | 73 | | 61 | 27 | 76 | | 53 | 43 | 40 | 43 | 12 | 40 | 47 |
| 95-M2 | 10 | | 53 | 18 | 51 | | 47 | 28 | 14 | 20 | 4 | 38 | 28 |
| 95-J2 | 10 | | 44 | 12 | 44 | | 31 | 15 | 7 | 20 | 2 | 26 | 21 |
| 95-A | 10 | | 19 | 0 | 0 | | 17 | 9 | 0 | 16 | 2 | 0 | 7 |

* Stage of development for the pistillate flowers, as indicated by month and growing season:
F = flowers; A = acorns; J, S, and M = July, September, and May, respectively;
1 and 2 indicate the first and second growing seasons.

Average survival of flowers to acorns is 10%.

Table 4. Variation in percentage of flower survival and acorn production in white oak

| Stage* | Tree No. 1 | 2 | 3 | 4 | 6 | 7 | 8 | 9 | 11 | Mean** |
|---|---|---|---|---|---|---|---|---|---|---|
| 90-F | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| 90-J | 27 | 18 | 31 | 5 | 23 | 12 | 15 | 22 | 55 | 25.9 |
| 90-A | 8 | 6 | 2 | 0 | 4 | 0 | 1 | 3 | 32 | 7.0 |
| 91-F | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 0 | 100 |
| 91-J | 37 | 35 | 25 | 0 | 27 | 5 | 23 | 3 | 0 | 17.6 |
| 91-A | 9 | 25 | 19 | 0 | 8 | 1 | 2 | 0 | 0 | 6.4 |
| 92-F | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| 92-J | 0 | 37 | 5 | 1 | 12 | 4 | 2 | 17 | 35 | 10.5 |
| 92-A | 0 | 13 | 2 | 1 | 6 | 0 | 0 | 3 | 24 | 5.0 |
| 93-F | 100 | 100 | 0 | 0 | 100 | 100 | 100 | 100 | 100 | 100 |
| 93-J | 16 | 30 | 0 | 0 | 12 | 12 | 17 | 15 | 39 | 18.1 |
| 93-A | 0 | 30 | 0 | 0 | 7 | 4 | 6 | 2 | 30 | 9.3 |
| 94-G | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| 94-J | 1 | 1 | 0 | 0 | 8 | 4 | 1 | 0 | 0 | 1.3 |
| 94-A | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0.4 |
| 95-F | 100 | 100 | 0 | 100 | 100 | 100 | 100 | 100 | 100 | 100 |
| 95-J | 65 | 43 | 0 | 17 | 49 | 40 | 44 | 26 | 66 | 42 |
| 95-A | 9 | 20 | 0 | 8 | 17 | 14 | 19 | 15 | 25 | 15 |

* F = flowers, J = flower survival on July 1, A = acorns.
** Average survival of flowers to acorns is about 7 percent.

Table 5. Correlation coefficients of individual black oak flower crops with their subsequent developmental stages during the two growing seasons

| Flowering year | Stage and growing season J1 | S1 | M2 | J2 | AC |
|---|---|---|---|---|---|
| 90-F | 0.940* | 0.935* | 0.927* | 0.921* | 0.429 |
| 91-F | 0.976** | 0.972** | 0.970** | 0.952* | 0.780 |
| 92-F | 0.911* | 0.892* | 0.847 | 0.731 | 0.631 |
| 93-F | 0.895* | 0.913* | 0.861 | 0.801 | 0.260 |
| 94-F | 0.724 | 0.724 | 0.692 | 0.649 | 0.232 |
| All years-F | 0.966** | 0.939* | 0.931* | 0.962** | 0.709 |

*, ** = P> 0.05 and 0.01, respectively (n = 12).

Table 6. Correlations of black oak flower development stages with each other for the flowering years 1990 to 1994

| | J1 | S1 | M2 | J2 | AC |
|---|---|---|---|---|---|
| J1 | --- | 0.980** | 0.792 | 0.741 | 0.955* |
| S1 | | --- | 0.825 | 0.748 | 0.879* |
| M2 | | | --- | 0.974** | 0.638 |
| J2 | | | | --- | 0.619 |
| AC | | | | | --- |

*, ** = P > 0.05 and 0.01, respectively. (n = 5)

Maximum temperature during pollination season (EPD-2 to EPD+1) was negatively correlated with black oak flower survival at the J1 and S1 stages (r = - 0.907* and - 0.950*, respectively). In white oak, maximum temperature was also negatively correlated with flower survival in July (r = - 0.950**) and with acorn production (r = -0.851*). Average temperature or days with fog, an indirect measure of humidity, showed no significant correlations with any developmental stages of either the white or black oak flowers.

Rain during the EPD was negatively correlated (P> 0.811) with white oak flower survival in July (r = - 0.829*). For black oak, there were negative correlations (r values between - 0.706 and - 0.837, ns) for rain during EPD+1 in relation to flower survival at all stages of development.

## DISCUSSION

Certain components of weather (days with hail and maximum temperature) during the pollination period were found to be negatively correlated with acorn production. Hail (Table 7) may potentially affect flower survival and acorn production in several ways: it may (1) damage or remove leaves, which reduces total leaf area and, thereby, reduces production of photosynthates for export to the developing acorns; (2) remove the fragile catkins (and, thus, the pollen source) if the hail occurs before shedding; and (3) damage or remove pistillate flowers, especially those of white oaks which, because of the long and slender peduncle, tend to be more fragile than those of black oak, which are tightly appressed to the branch. However, the long styles of the black oaks are probably more susceptible to hail damage than the sessile styles of white oaks. It is not unusual to observe black oak flowers in their first or second growing season with missing styles. If styles are broken off before pollen tubes cease growing in mid-May (Cecich and Haenchen 1995; Cecich, in press), then the flower will not be fertilized in the second growing season and will abort. Since pollen tubes cease growth just below where the style fractures (the visible portion of the perianth (Cecich, in press)), they have the potential for resuming growth to accomplish fertilization.

The negative correlation of maximum temperature during the pollination season for black and white oak flower survival at J1 may indicate that the elevated temperature put pollen development out of synchrony with the receptivity of the pistillate flowers, causing the females to abort from lack of pollen. For instance, Sharp and Sprague (1967) speculated that a warm late April and a cool early May were related to early catkin emergence and delayed pollen dispersal, respectively. At their site, pollen usually was shed in late May. Under those weather conditions, pollen shed and pistillate flower receptivity were considered to be more closely aligned and thus related to good acorn crops. Similarly, in the current study, with pollen shed usually occurring in late April or early May, the relatively dry and warm conditions for EPD-1 and EPD-2 in 1994 may have altered the time relationship between pollen shed and pistillate flower receptivity, especially if considered in conjunction with the lower temperature and higher relative humidity during the EPD (Table 6). In white oak, maximum temperature was also negatively correlated with acorn production, which is highly correlated to flower survival at J1.

Table 7. Weekly averages of weather variables in relation to the EPD classes for black and white oak

| EPD's | DATE | | Weather variables AVE C | MAX C | HUM | FOG | RAIN | HAIL |
|---|---|---|---|---|---|---|---|---|
| EPD-2 | 4/23/90 | | 20.8 | 27 | 68 | 3 | 0 | |
| | 4/23/91 | | 11.3 | 27 | 71 | 1 | 1 | |
| | 4/9/92 | | 14.7 | 22 | 61 | 3 | 2 | |
| | 4/23/93 | | 12.5 | 19 | 68 | 4 | 2 | |
| | 4/16/93 | a | 10.0 | 15 | 82 | 4 | 1 | 2 |
| | 4/16/94 | | 16.6 | 25 | 51 | 0 | 1 | 1 |
| | 4/16/95 | | 14.4 | 20 | 64 | 3 | 3 | |
| EPD-1 | 4/30/90 | | 12.4 | 17 | 72 | 2 | 1 | |
| | 4/30/91 | | 16.5 | 17 | 65 | 2 | 4 | |
| | 4/16/92 | | 18.0 | 23 | 71 | 3 | 3 | 1 |
| | 4/30/93 | | 17.5 | 22 | 78 | 2 | 0 | |
| | 4/23/93 | a | 12.5 | 19 | 68 | 4 | 2 | |
| | 4/23/94 | | 17.4 | 24 | 55 | 2 | 3 | 1 |
| | 4/23/95 | | 11.1 | 17 | 68 | 3 | 2 | |
| EPD | 5/7/90 | | 14.2 | 21 | 69 | 3 | 1 | |
| | 5/7/91 | | 16.6 | 21 | 73 | 4 | 1 | |
| | 4/23/92 | | 9.9 | 15 | 74 | 1 | 2 | |
| | 5/7/93 | | 20.2 | 26 | 78 | 0 | 2 | |
| | 4/30/93 | a | 17.5 | 22 | 78 | 2 | 0 | |
| | 4/30/94 | | 10.0 | 15 | 72 | 3 | 2 | 1 |
| | 4/30/95 | | 9.9 | 14 | 75 | 5 | 0 | |
| EPD+1 | 5/14/90 | | 15.3 | 21 | 78 | 6 | 1 | |
| | 5/14/91 | | 23.5 | 20 | 76 | 5 | | |
| | 4/30/92 | | 16.8 | 23 | 58 | 1 | 0 | |
| | 5/14/93 | | 16.2 | 22 | 71 | 0 | 2 | |
| | 5/7/93 | a | 20.2 | 26 | 78 | 0 | 2 | |
| | 5/7/94 | | 15.2 | 21 | 72 | 2 | 1 | |
| | 5/7/95 | | 14.7 | 19 | 80 | 6 | 2 | |

a. The EPD dates and associated weather data for black oak in 1993 are 1 week earlier than those of white oak.

The negative correlation for white oak acorn production in the previous year and flower survival in July ($r = -0.909$) may indicate that storage reserves in the tree are critical to the acorn development process only until the time of fertilization in mid-June. Then, as the acorn grows, primarily during cotyledon enlargement, the current-year photosynthates may become important as the energy source. Kossuth (1974) found that developing leaves and pistillate flowers of white oaks were not competitive sinks for photosynthates. The positive correlation of July flower survival with acorn production 2 years prior ($r = 0.913$) and the negative correlation for the previous year ($r = -0.909$) can be viewed as evidence for biennial-bearing in white oak. However, this view must be tempered with the fact that, because of the number of years of data, the threshold for a significant r-value is 0.950. Sork and others (1993) provided evidence of a 3-year fruiting cycle in white oak and a 2-year cycle in black oak by using 8 years of individual tree data. Since there is little loss of white oak flowers between July and September, the July 1 value appears to be a good estimator of current-year acorn production.

There are two possible causes for the high flower counts in 1990: (1) an increase in the number of peduncles with 3 pistillate flowers (normally, the black oaks in this population have 2 pistillate flowers per peduncle; a few have 1 or 3); (2) an increase in the number of leaves per branch with flowers in their axils. That is, those leaves that usually would have vegetative buds in their axils could have had pistillate flowers. This assumes there was no increase in the number of leaves per branch in 1990. Unfortunately, a record of this potential variation was not kept during the study.

Genetic control over seed production has been examined by numerous investigators (Cecich 1993). For instance, Farmer (1981) found that in a given year acorn production in white oak was highly correlated with the percentage of fertilized pistillate flowers, while year-to-year differences were associated with variation in the number of flowers. The current study also provides evidence of tree-to-tree variation in white and black oak for both flower production and acorn production (Tables 1, 2). But, the data also suggest that weather factors may mediate the pollination process and eventually affect fertilization and acorn development. The coefficients of variation for flower production of the two species are identical (Tables 1, 2), but the twofold difference in cv's for acorn production indicates that factors such as weather are affecting subsequent development. The difference in cv's for acorn production in the black and white oaks (42% and 87%, respectively) also reflects the year-to-year consistency of acorn production in black oak (Sork and Bramble 1993).

Since the visible portions of the reproductive cycles in white and black oaks require one and two growing seasons, respectively, can one realistically ascribe the success or failure of the development of pistillate flowers into acorns to factors such as weather? That is, how and when does the weather affect differentiation? Although the data were evaluated differently, the present findings support the position of Sork and others (1993) and Koenig and others (1994a) that weather and previous reproductive history affect seed production. Furthermore, the results of this study provide evidence that weather at the time of a specific developmental event, such as pollination, can be correlated to acorn production.

## LITERATURE CITED


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# MODEL OF WHITE OAK FLOWER SURVIVAL AND MATURATION

David R. Larsen[1] and Robert A.Cecich[2]

Abstract: A stochastic model of oak flower dynamics is presented that integrates a number of factors which appear to affect the oak pistillate flower development process. The factors are modeled such that the distribution of the predicted flower populations could have come from the same distribution as the observed flower populations. Factors included in the model are; the range of peak crop intervals, relative humidity in the pollination period (Wolgast 1972), number of severe storms (Cecich 1997), tree hopper (Kopp and Yonke 1973a, 1973b) and acorn weevil population (Gibson 1964), genetic factors related to the fertilization process, and summer drought (Sork and Bramble 1993).

We propose this probabilistic model as a research tool to explore how the included factors affect acorn production. Because the model uses distributions similar to those obtained from bootstrap analysis of the observed data, realizations of the model are generated using a stochastic modeling methodology and then summarized for presentation. With this methodology, the output will be drawn from the distributions similar to the original data and fit statistics are relatively meaningless.

## INTRODUCTION

Flower development in oak trees is the defining process in determining the abundance of the acorn crop in a particular year. A large amount of research has been conducted on insect damage to developing acorns (Kearby and others 1986), relatively less attention has been focused on the size of the flower crop and the steps in the pollination and maturation process that may limit acorn crop size (Cecich 1993). Only a few studies have monitored annual oak flower production over a long time frame (Sharp and Sprague 1967; Sork and others 1993; Sork and Bramble 1993; Koenig and others 1994, 1996). Of these, Koenig's study is 16 years long, Sharp and Sprague's study extended 11 years, and Sork's data set covers 9 years. Only Koenig's study and the data used in this study are continuing to collect data. These studies and others indicate that the process of oak flower development is highly stochastic and influenced by both tree specific flower production, average climatic trends, specific weather events and insect populations. Because the process is highly variable and only one observation is added annually, we developed a model of the hypothesized processes to explore its consequences. Correlation analysis indicated a number of factors that are negatively correlated with abundance of oak flower crops at various times (Sork and Bramble 1993, Cecich 1997). Our model describes how these factors affect overall oak flower abundance and tests the predictions against empirical data.

Because the various species of oak have a variety of phenologies, white oak (*Quercus alba* L.) was chosen for the initial model building process. White oak completes the flower to acorn cycle within a single year, an advantage for developing this model. Additionally, our sample dataset contains a number of white oak trees that vary in flower populations and acorn crop sizes.

In this paper, we present a model of the white oak flower development process as four major steps: size of the pre-pollination flower population, May and June survival, June and July survival, and the July and September survival. The hypothesized distribution of flower survival reflects the distribution of observed flower survival in the original sample. These hypothesized distributions are modeled as a function of the dominant factors that can affect flower survival, as identified by the authors above.

[1] The School of Natural Resources, University of Missouri-Columbia, 1-30 Agriculture Building, Columbia, MO 65211, dlarsen@sylvan.snr.missouri.edu.

[2] USDA Forest Service, North Central Forest Experiment Station, Columbia, Missouri 65211.

## MODEL AND EMPIRICAL DATA

The model is designed with two key considerations: (1) it must accurately reflect the distributions in the observed data and (2) it must provide a flexible method of evaluating alternative hypotheses affecting those distributions. The model follows the general form of first generating a hypothetical population of white oak flowers at the start of the pollination period. Then the survival of oak flowers is predicted for each of three time periods, May-June, June-July, July-September. The survival in each of these periods is influenced by a variety of factors that can reduce the population of oak flowers. The proposed causes of oak flower survival are not an exhaustive list of causes, just those that reflect the author's view of the oak flower survival and maturation process. Figure 1 diagrams the components of the model and the factors that effect each.

$$n_t = \left( \left( \left( \beta(v,w) \cdot \max \right) \cdot \frac{\sum p_i}{n_{p_i}} \right) \cdot \sqrt[n_{p_j}]{\prod p_j} \right) \cdot \frac{n}{\sum \frac{1}{p_k}}$$

where $n_t$ is the number of oak flowers that will survive and mature into acorns in September for that year, $\beta(v,w)$ is a beta distribution with parameters $v$ and $w$ which describe the shape of the beta distribution. If the parameters are the same value the distribution is symmetrical, as the values increase kurtosis increases and as the $w$ increases relative to $v$ the distribution skews to the left. The values generated are between 0 and 1. The constant max in the maximum potential flower crop per tree sample. Variable $p_i$, $p_j$, and $p_k$ are the proportional reduction by the various factors in the model at times $i$, $j$, $k$, which correspond to May-June, June-July, and July-September time periods. While this model seems quite complicated it is really quite simple, we will describe each part separately to clarify the purpose of each component.

The model must first generate an oak flower population for the year at the start of the pollination period. We chose to start at this point because the processes governing flower initiation are much less distinct and harder to quantify. The pollination period for the study data is typically a 1 week period in late April or early May. This period can vary by a week earlier in some years and a week later in other years. The number of flowers for a particular year varies greatly but tends to have a cyclic pattern. The probability of a large number of white oak flowers in a year decreases after a peak year. The probability increases as the number of years after a peak year increase. This relationship is described by the following equations:

$$n = \beta(v,w) \cdot \max$$

In this equation, $v$, $w$ are parameters of a beta distribution ($\beta$) and max is the maximum flower sample taken in manner of Cecich, (1997). The parameters $v = 3.0$ and $w = 9.0$ were used in this study as they approximate the distribution of initial flower populations determined from a bootstrap analysis of the observed data.

The second component of the model predicts the proportional reduction in the flower population given causes that happen within the period May-June. This period includes factors that affect the pollination process. We model three causes for flower reduction in the May-June period; pollination weather (relative amount of time below 60% relative humidity during the 1 week pollination period), favorable weather (1 - relative frequency of severe storms that cause flower damage (e.g, hail storms or sever winds)), 1 - (the relative number of tree hoppers (*Membracidae*)). All these measures are relative (values between 0 and 1) with 1 being a positive condition for oak flowers. For this component, the arithmetic mean of the three components produce the proportional reduction in the oak flower population. It was found that the original frequency distribution had little or no skewness and this mean reflects that property. The model is:

$$s_1 = \frac{1}{n} \sum_{i=1}^{n} p_i$$

where $n$ is the number of factors being averaged (3 in this case) and $p_i$ is the proportion remaining for each factor.

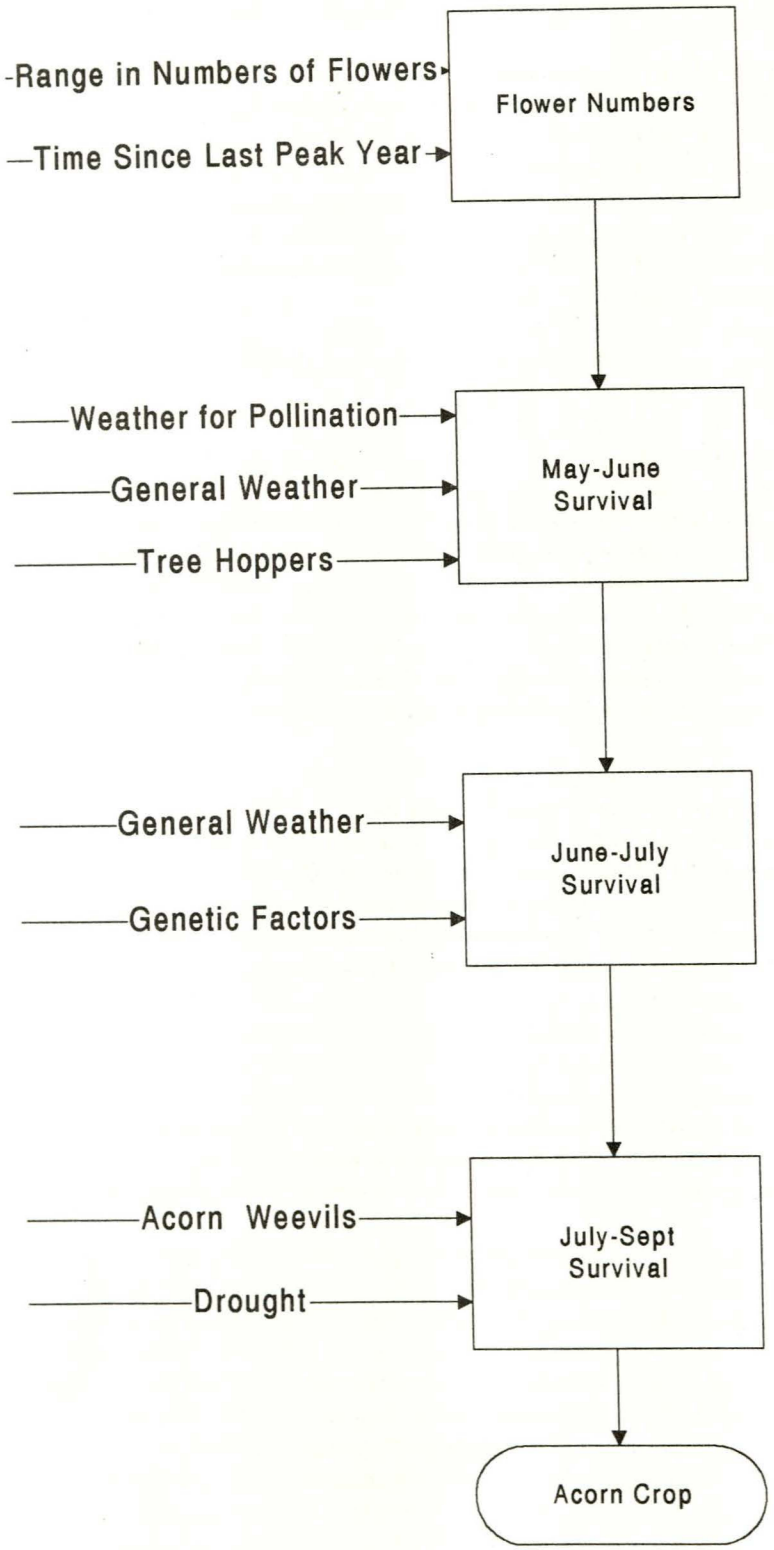


Figure 1. Diagram for the component of the oak flower development model and the factors affecting each component. The distributions in each box are constrained to produce distributions similar to those observed in Missouri white oaks over a 6 year period.

The third component of the model describes the June-July survival and is hypothesized to be driven by two factors; favorable weather in this period, and genetic factors relating to the fertilization process (Cecich, 1997). The favorable weather component is again 1 - (the relative frequency of severe storms that cause flower damage). Genetic factors relate to the success of the fertilization process. This is the period in which fertilization occurs and the growth and maturation of the acorn starts. Lethal gene combinations would disrupt the embryogenic process. These factors are combined by taking the geometric mean. This mean can be skewed to the left of the proportional range similar to the observed data at this time period. The equation is:

$$s_2 = \sqrt[n]{\prod_{j=1}^{n} p_j}$$

where $n$ is the number of factors (2 in this case) and $p_k$ is the proportion remaining for each factor.

The fourth component of the model described the survival of flowers/acorns from July to September and includes two traditional causes of acorn population reduction: acorn weevils and summer drought. Again, we changed these factors so that a value of 0 indicates poor survival for oak flowers/acorns and a value of 1 indicates the best survival of the flowers/acorns. In this component, the harmonic mean of the relative proportions is used to describe the survival as this produces a highly skewed distribution similar to the observed frequencies at this time period. The equation for the third component is as follows:

$$s_3 = \frac{n}{\sum \frac{1}{p_k}}$$

where $n$ is the number of factors (2 in this case) and $p_k$ is the proportion remaining for each factor.

These three type of means used in this study have the property that the harmonic mean ≤ geometric mean ≤ arithmetic mean. If the proportions are equal all three means have the same value otherwise each of the means is skewed to the left as indicated. In terms of the model, the negative effect of several factors is the arithmetic mean of the factors in the first component, with each successive component the mean negatively skewed relative to the arithmetic mean except in the case there the proportions are equal in which case the means are equal.

This model was calibrated using a data set fromnine white oak trees on the Baskett Wildlife Area in central Missouri. Each tree was sampled weekly through the flower development period April-June and then periodically for the remainder of the season. Five branch ends (the past 2 years) are sampled on each tree from the top and sides of the crown. These data were collected since 1990. The year 1996 was not included in the analysis as the data collection was not completed at the time of analysis. The data are summarized as average per tree values for all trees in the stand. A more complete description of this data can be found in this volume (Cecich 1997). The model used the tree sample as the unit of estimation.

## RESULTS AND DISCUSSION

Because the model was built in components that matched the empirical observations, each component was designed to allow a varying number of input factors while maintaining the general distributions observed in the data set. The behavior of this model is constrained to the general range of the empirical data.

The model of flower populations is designed to be a highly stochastic. The estimated pattern is assumed to be the combination of two factors. Annual variation in the general weather pattern, such as sufficient moisture and the presence or absence of violent storms with the cyclic pattern is assumed to be due to the energy required to produce a large acorn crop provide the basis for the stochastic effects. Figure 2 illustrates the behavior of these equations for a 100 year period. This prediction assumed a average of 50 flowers per tree sample, and a maximum of 400 flowers per tree sample (an extremely good year).

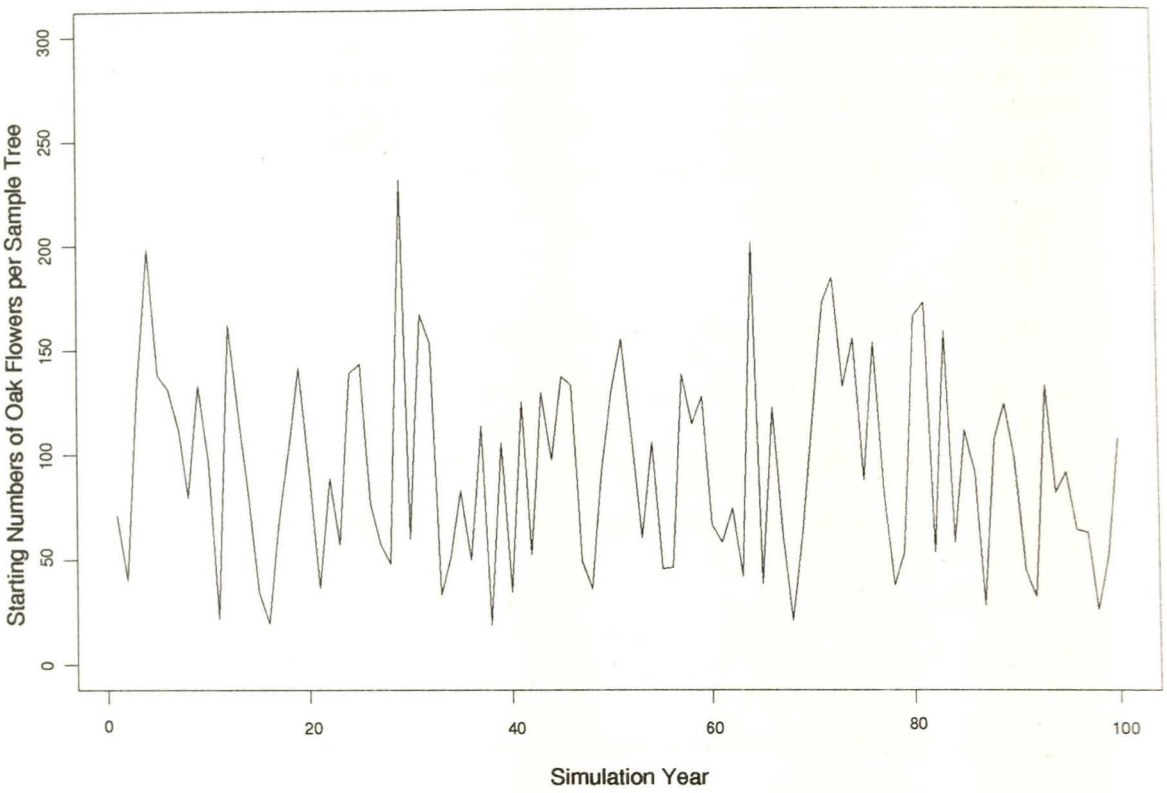


Figure 2. The predicted white oak flower population for a 100 year period using the parameters listed in the paper. The numbers on the y axis indicate the number of oak flowers per five samples of 2 year old branch units.

The values produced in this manner can be considered to come from the same population as the observed data, however, because the observed data is only for 6 years, it is difficult to say that this model completely describes the process. This method of modeling may not be familiar to many of the readers, the intent is to describe a distribution from which the observed data *could have come*. The true distribution will remain unknown until all possible white oak trees are measured for their entire life span. Additionally the distributional approach does not attempt to predict the exact number of flowers in a particular year as in deterministic models. Rather the intent is to describe the probable frequency distribution of possible outcomes over a long period of time.

The three mean components of the model produce proportional reductions to the flower population that parallel the observed distributions. Because the factors are considered independent, large reductions can occur in one period with very small reductions in the next period. On average in the first period oak flowers are reduced by 50%, the second period the remaining flowers are reduced by an additional 35% and the third period the remaining flowers are reduced by another 11%.

To illustrate the overall behavior of the model, 100 simulations were generated to illustrate the range of outcomes. Figure 3 summarizes these simulations with maximum and minimum lines as well as the average line for the simulations. The observed data is presented for reference. Note that in most years, only modest numbers of flowers are initiated, which are further reduced through the season. There are some years in which large numbers of oak flowers are initiated. In some of those years the flower populations are severely reduced and in other years they are not.

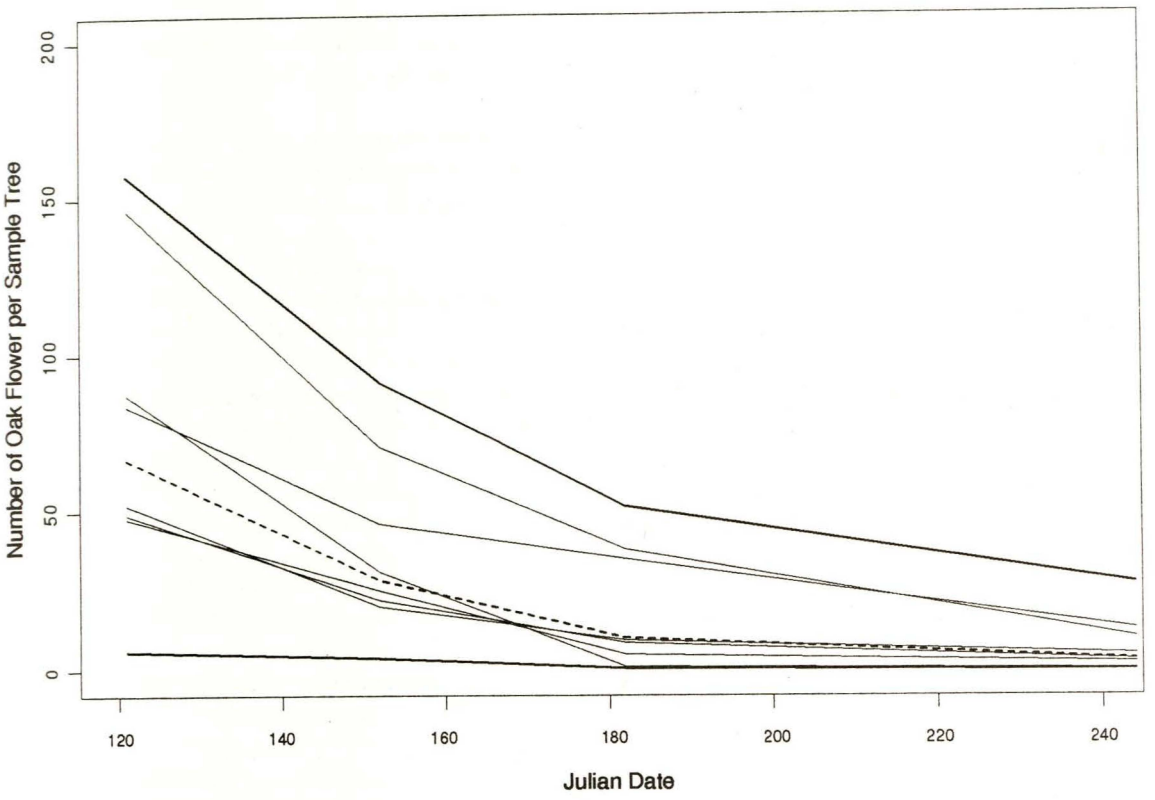


Figure 3. The predicted white oak acorn development process as illustrated by the model. 100 simulations are summarized as the maximum and minimum values (double weight solid lines), the smoothed average of the simulations (double weight dashed line) to illustrate the range of outcomes predicted by the model. The observed data is includes for reference (single weight solid lines). The numbers on the y axis indicate the number of oak flowers per five samples of 2 year old branch units.

## CONCLUSION

The proposed model generates distributions of white oak flower populations through a growing season that have the similar distributions, at several intermediate points, to the observed data. These distributions incorporate factors suggested by others to effect changes in flower/acorn populations. While it is not possible to determine that this model is the "correct" characterization of the process, it is possible to use the model to explore the consequences of change in the included factors. If it is assumed to contain major factors affecting the process, the model becomes a useful tool to explore the limits of these factors in explaining the variations in white oak acorn crop production.

This model integrates background periodicity of the acorn production process, weather at specific periods in the pollination, fertilization and maturation process, and other factors that decrease potential acorn crop size. Weather factors that are considered to be important include the amount of time below 60% relative humidity during the pollination period, the number of severe storms with high winds or hail during the growing season, and summer drought that may effect maturation. Other factors include the size of the tree hopper population in the June-July period and the size of the oak weevil population in late summer.

This model could provide a valuable research tool for exploring the consequences of assumed changes in the factors affecting acorn production.

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# CORRELATIONS AMONG STAND AGES AND FOREST STRATA IN MIXED-OAK FORESTS OF SOUTHEASTERN OHIO

P. Charles Goebel[1] and David M. Hix[2]

Abstract: Many models of landscape ecosystem development, as well as of forest stand dynamics, are based upon spatial and temporal changes in the species composition and structure of various forest strata. However, few document the interrelationships among forest strata, or the response of different strata to alterations of natural disturbance regimes. To examine how relationships among forest strata varied with stand age, we sampled a chronosequence (five age classes) of 21 second-growth (70 to 149 years old) and old-growth (≥ 150 years old) mixed-oak stands from a single ecosystem type within the ecological classification system framework for the Wayne National Forest. All woody stems were measured by dbh and species, and classified into these strata by lifeform: (1) canopy (dominant, codominant, and intermediate crown classes); (2) saplings (stems < 10 cm dbh, but > 1.37 m tall with canopy potential); (3) subcanopy (stems < 10 cm dbh, but > 1.37 m tall without canopy potential); and (4) seedlings (stems < 1.37 m tall with canopy potential). Canonical correspondence analysis (CCA) was used to summarize the compositional variation in each strata by age class. Canonical correlation analysis was then used to examine the interrelatedness among forest strata using CCA axis scores. CCA ordinations revealed several trends, primarily that site differences and stand age were important factors constraining the development of various forest strata. Although there was no significant correlation between the canopy and sapling strata, there was strong correlation between the canopy and seedling strata, as well as between the sapling and seedling strata. These canopy, sapling, and seedling strata also were strongly correlated with their canonical variables, suggesting that each is responding to similar gradients. We suggest that the natural disturbance regimes of the study area have been altered, and as a result, the compositions of the mixed-oak stands in southeastern Ohio are responding to these modifications. Whereas a primary landform effect regulating landscape ecosystem and stand development prior to European settlement was likely the spatial pattern of surface fires, the effect of landform on the frequency and spatial geometry of these surface fires has diminished with fire suppression efforts.

## INTRODUCTION

Models of landscape ecosystem development are predicated on a hierarchical structure, with the upper levels of the hierarchy constraining the lower levels (Figure 1). At broad continental scales, climatic and large-scale physiographic variation mediate the development of regional landscapes (Bailey 1996). Within these regional landscapes, differences in geology and physiography, as expressed by landforms, reflect the differences in the type of geologic surface deposits and depositional environments. Landforms, in turn, control the local microclimate, soil genesis, and local disturbance regimes (Rowe 1984, Host and others 1987, Bailey 1996, Goebel and others 1996). These constraints result in distinctive and repeatable patterns of vegetative composition, structure, and development (Barnes and others 1982, Hix and Lorimer 1991, Goebel and Hix 1996).

Swanson and others (1988) suggest that there are four major effects that landforms have on the patterns and processes of landscape ecosystem development. First, landforms (by their elevation, aspect, parent material, and slope steepness) influence the local climatic regimes and the levels of moisture, nutrients, and other materials that are available to landscape ecosystems. Second, landforms affect the flow of organisms, propogales such as seeds and sprouts, energy, and other materials (e.g., organic matter, water) through a landscape ecosystem. Third, landforms can influence the frequency and spatial pattern of non-geomorphic disturbances, such as fire, wind, and grazing, that are expressed through changes in the composition and structure of biotic communities. Finally, landforms constrain

[1] Research Technician, Joseph W. Jones Ecological Research Center, Route 2, Box 2324, Newton, GA 31770.
[2] Assistant Professor, School of Natural Resources, The Ohio State University, 2021 Coffey Road, Columbus, OH 43210-1085.

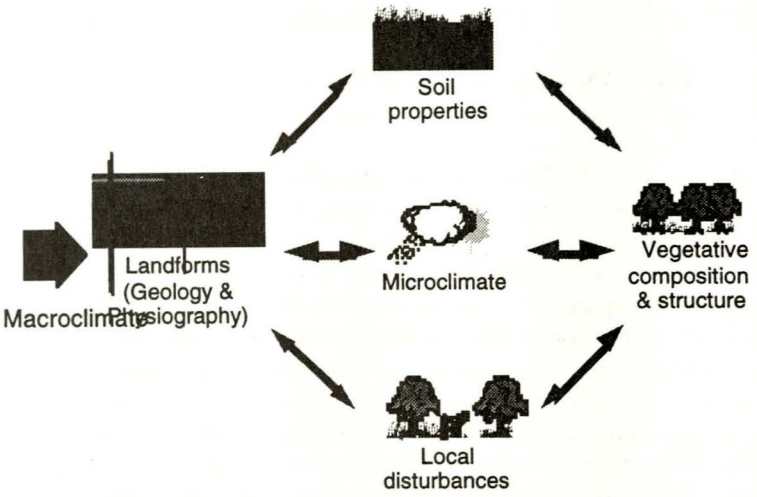


Figure 1. Hierarchical controls on landscape ecosystem development. The left side represents the upper levels of the hierarchy, while the right side represents the lower levels of the hierarchy. Modified from Bailey (1996) and Rowe (1984).

the spatial pattern and rate or frequency of geomorphic processes that alter biotic features and processes. When disturbance regimes are suppressed or altered, the interaction of these landform effects on ecosystem development may influence strongly the persistence and geometry of the mosaic of landscape ecosystems (Swanson and others 1988).

Multifactor and hierarchical ecological classification systems that base the higher levels of the system on landforms, such as a recent classification developed for the Wayne National Forest (Hix and Pearcy 1996), provide a framework for separating and identifying landscape ecosystems (Barnes and others 1982, Bailey 1987). These hierarchical classification systems also provide an opportunity to examine the effects of landforms on the development of an individual landscape ecosystem type. By examining a single ecosystem type, the landform effects on environmental gradients and movement of materials (class 1 and 2 effects; Swanson and others 1988) are assumed to be constant, allowing for analysis of the landform effects on disturbance regimes (class 3 and 4 effects).

The objective of this study was to examine the landform effects on the successional mechanisms that may be controlling the later-stage development of south-facing, upper to middle slope, second-growth and old-growth forest ecosystems in southeastern Ohio. Analysis of the development mechanisms were accomplished by: (*i*) quantifying the relationships between various strata and stand age on a single ecosystem type through the use of a chronosequence, and (*ii*) determining whether the various forest strata were highly correlated on a single ecosystem type, thus providing information on the inter-stratal relationships among various stand ages. Our hypothesis is that highly correlated strata on a single ecosystem type are responding to the same environmental conditions (class 1 and 2 effects), and thus any compositional differences are most likely responding to differences in stand age and (or) disturbance patterns (class 3 and 4 effects).

## STUDY AREA

The current study was conducted on public and private lands located in southeastern Ohio (Figure 2) within the Western Hocking Plateau Subsection (221Ef) of the Southern Unglaciated Allegheny Plateau Section (221E) in the Eastern Broadleaf Forest Province (221; Keys and others 1995). The climate of the area is humid continental with relatively cold winters and warm summers. Mean daily temperatures are 0° C and 21° C , respectively (Lucht and others 1985). Precipitation averages 98 cm annually, of which half falls during the growing season (Lucht and others 1985).

The Western Hocking Plateau Subsection is a dissected plateau with moderate to steep slopes, narrow ridgetops, rock outcrops, and narrow stream valleys with elevations ranging from 195 to 322 m above sea level. Bedrock geology consists of inter-bedded sandstone, shale, siltstone, limestone, and coal (Rympa 1961, Keys and others 1995). In general, the forest soils are moderately acidic and were formed in colluvium (Keys and others 1995). Specifically, the dry upper to middle hillslopes (ecosystem type or ecological landtype phase (ELTP) 32) examined in the current study are south-facing, tend to be moderately to steeply sloping, and have moderately deep and well-drained soils with loamy over clayey textures (Hix and Pearcy 1996).

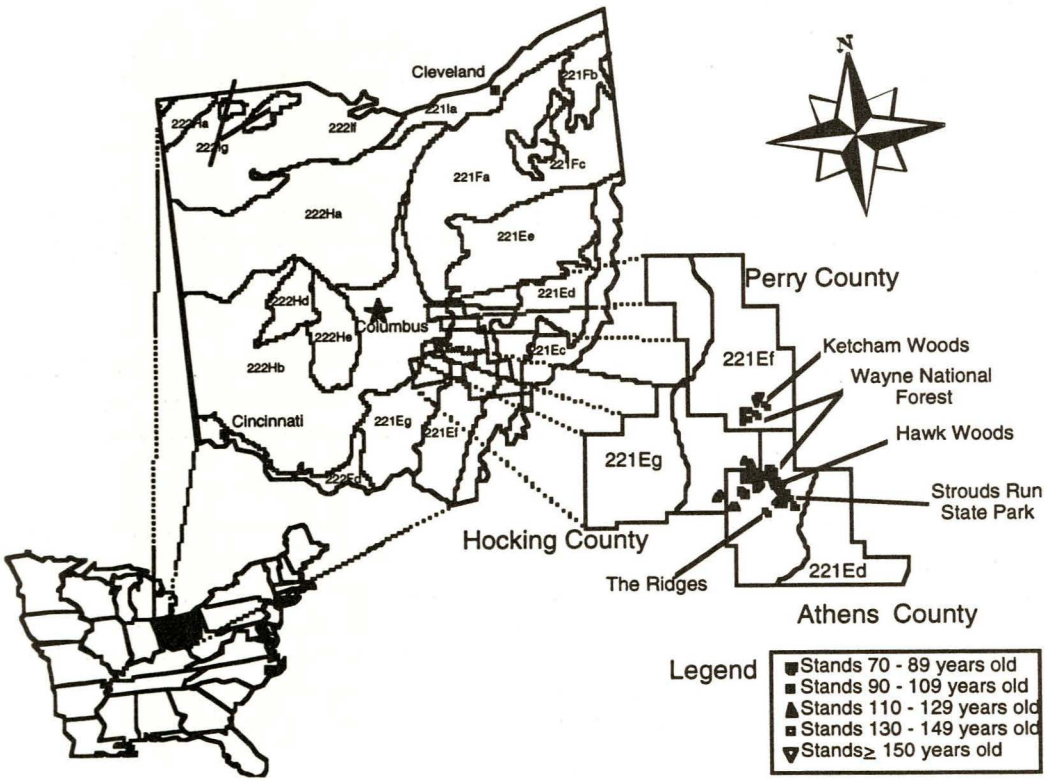


Figure 2. Location of second-growth and old-growth study areas within the Western Hocking Plateau Subsection (221Ef) of southeastern Ohio.

In southeastern Ohio, frequent burning is believed to have historically favored the more fire-resistant oaks, and eliminated understories of mesic species, such as American beech (*Fagus grandifolia* Ehrh.) and sugar maple (*Acer saccharum* Marsh.), resulting in a disturbance climax (Beatley 1959, Rympa 1969). Gordon (1969) suggested that the presettlement vegetation types of the study area were mixed-oak, mixed-mesophytic, and beech forests. However, the presettlement forests on south-facing ecosystem types in southeastern Ohio were most likely oak-hickory or mixed-oak forests (Cutler 1812, Boetticher 1929, Beatley 1959, Rypma 1961, Goebel and Hix 1996). The second-growth forests of southeastern Ohio have been heavily disturbed by agricultural conversion, timber harvesting, coal mining, and livestock grazing. Currently, oak-hickory and Appalachian mixed-hardwoods (Eyre 1980) that established after extensive clearcutting approximately 70 to 140 years ago are the dominant second-growth forest types of the study area (Ducey 1982).

The south-facing, upper to middle slopes examined in this study are dominated by white oak (*Quercus alba* L.). Canopy white oak importance has been shown to be significantly greater in older second-growth stands than younger second-growth stands (Goebel and Hix 1996). However, the canopies of old-growth ecosystems on similar south-facing, upper to middle slopes were dominated by mixed-oaks, including white oak, black oak (*Q. velutina* Lam.), chestnut oak (*Q. prinus* L.), and scarlet oak (*Q. coccinea* Muenchh.). Additionally, the sapling strata of both the second-growth and old-growth south-facing, upper to middle slopes were dominated by shade-tolerant species, such as red maple (*Acer rubrum* L.) and sugar maple, while the seedling strata of the older second-growth and the old-growth stands were comprised of shade mid-tolerant oak species (Goebel and Hix 1996).

## METHODS

During the summers of 1993 and 1994, 21 forest stands were sampled on the Athens Unit of the Wayne National Forest and adjacent public and private property. The sampling design consisted of a sequence of stands of different ages (a chronosequence) at least 70 years old. The chronosequence included stands of the following five age classes: 70 - 89, 90 - 109, 110 - 129, 130 - 149, and ≥ 150 years old. Because old-growth forests are rare in southeastern Ohio and those present are generally believed to be less than 250 years old (Davis 1993, McCarthy 1995, Goebel and Hix 1996), the old-growth age class (≥ 150 years old) did not represent a specific age range (e.g., 150 - 169 years old). Rather, the old-growth age class provides a general comparison of the forest conditions at the time of European settlement. Specific details on the procedures used to verify the chronosequence can be found in Goebel and Hix (1996).

Within each identified stand, two plots were established randomly on a transect that bisected the area along the contour; the first plot located randomly between 20 and 30 meters from the stand boundary, and the second plot installed randomly at least 40 to 50 meters from the first plot. Each sample plot consisted of a set of two concentric, circular subplots (sizes = 500 $m^2$ and 100 $m^2$), and eight rectangular 1- by 2-m quadrats. The centers of the quadrats were located 2 m beyond the outside edge of the 100 $m^2$ subplot in eight directions (N, NE, E, SE, S, SW, W, NW). On each 500 $m^2$ plot the species, dbh (diameter at breast height, 1.37 m) and crown class (dominant, codominant, intermediate, or overtopped, c.f. Smith 1986) of all living trees > 10.0 cm dbh were recorded. Sapling and subcanopy individuals were measured by species and dbh on the 100 $m^2$ plots, while seedlings were counted by species on the 2 $m^2$ plots. Members of the sapling strata included those individuals ≤ 10.0 cm dbh but ≥ 1.37 m tall with canopy potential (Rheinhardt 1992), whereas subcanopy members were the same size but whose lifeform relegated them to understory or subcanopy positions. Seedlings were individual woody plants < 1.37 m in height with canopy potential. Scientific nomenclature follows Gleason and Cronquist (1991), and common names follow Little (1979).

Basal areas and densities of canopy trees, and densities of sapling and seedling species, were converted from a plot basis to per hectare basis. Relative density (percent) was calculated by summing the total density on a plot by species, dividing by the total density of all species on the plot, and multiplying by 100. Relative dominance (percent) of canopy species was similarly computed, substituting basal area for density. Importance values (IV, summation of relative density and relative dominance divided by 2) were calculated for canopy trees (dominant, codominant, and intermediate crown classes), while relative densities were calculated for each woody sapling, subcanopy, and seedling species.

Canonical correspondence analysis (CCA) was used to explore the variation in species composition among the various age groups and selected stand factors (CANOCO; ter Braak 1990). CCA was performed on four separate matrices: (*i*) canopy species, (*ii*) sapling species, (*iii*), subcanopy species, and (*iv*) seedling species. Monte Carlo permutation procedures were used to test the significance of each stand factor in each CCA (ter Braak 1990).

Canonical correlation analysis (COR; proc CANCORR; SAS 1989), a linear multivariate regression analysis that maximizes the correlation of two matrices, was used to measure the correspondence between pairs of datasets (Gittens 1985, Jongman and others 1995). An artifact of the regression analysis is that the number of species plus

the number of environmental factors must be smaller than the number of sampled stands (Jongman and others 1995). To account for this restriction, we used the CCA stand scores in the canonical correlation analysis of all possible ordination combinations ($n = 6$). COR generates three statistics that are instrumental in determining the correspondance between two datasets. These include: (*i*) the canonical correlation coefficient of each axis with each canonical variable; (*ii*) the percentage of variance extracted by the stand factors for each canonical variable; and, (*iii*) the redundancy, or proportion of the variation in a vegetative stratum (as summarized by CCA axes 1 and 2) accounted for by each canonical variable (Gittens 1985).

## RESULTS

### CCA Ordinations

The ordinations of all sample plots by strata illustrate the variation in woody species composition found in the second-growth and old-growth forest ecosystems on south-facing middle to upper slopes in southeastern Ohio. Although selected from a single ecosystem type, the stand factors combined only explain 64.6% of the variation in canopy IV along the first two canonical axes. The first canonical axis was not significantly correlated with stand age ($P = 0.16$), however, the first axis was positively associated with scarlet oak IV and shagbark hickory IV. The second CCA axis was positively associated with higher values of chestnut oak IV (Figure 3). These results confirmed that that the older, second-growth stands are characterized by higher IVs of longer-lived, and shade-tolerant white oak while the younger second-growth stands are comprised by shorter-lived, more shade-intolerant species (Table 2). Old-growth canopies were comprised of a mixture of oak species. While elevation was significantly correlated with the first axis ($P \leq 0.05$), the other stand factors were not strongly correlated with either canonical axis (Table 1), suggesting that the influence of the measured stand factors on canopy composition is minimal.

Whereas the canopy CCA was not highly constrained by stand age, the sapling CCA was significantly constrained by several stand factors, including stand age (Table 1). The sapling layers of most second-growth and old-growth stands were characterized by red maple (Table 2). However, sugar maple relative densities were highest in the 90 to 129 year old stands, while American beech saplings were more common in the old-growth stands (Table 2). The CCA triplot reveals that several stand factors significantly constrain the distribution of sapling species (Figure 3). Aspect, percentage of the distance to the ridgetop (PDR), percent slope, and stand age are all significantly correlated with the first CCA axis, while aspect and slope shape were significantly correlated ($P \leq 0.05$) with the second CCA axis (Table 1). This suggests that within a single ecosystem type, small differences in the measured stand factors may result in different sapling compositions. For example, older stands with steep, southwesterly slopes have higher relative densities of red maple and American beech.

Although the stand factors combined to explain 79.1% of the variation in subcanopy relative density (Table 1), stand age was not significantly correlated with either the first or second canonical axes ($P > 0.05$). Slope shape and aspect were both strongly correlated with both CCA axes ($P \leq 0.01$), suggesting that eastern hophornbeam (*Ostrya virginiana* (Mill.) K. Koch) relative density was higher on more xeric, southwesterly facing, convexed-shaped slopes, while serviceberry (*Amelanchier arborea* (Michx.f.) Fern.) and flowering dogwood (*Cornus florida* L.) relative densities were higher on more dry-mesic, southerly facing, concave-shaped slopes (Figure 3). Old-growth stands were centrally located along both the first and second CCA axes (Figure 3).

Table 1. Summary statistics of the canonical correspondence analyses for each strata.

| | CCA Axis 1 | CCA Axis 2 |
|---|---|---|
| *Canopy:* | | |
| Eigenvalue | 0.20 | 0.15 |
| Species-environment correlation | 0.69 | 0.61 |
| Cumulative % variance of species | 7.70 | 13.40 |
| Cumulative % variance of species-environment relation | 37.10 | 64.60 |
| Monte Carlo Permutation tests: | | |
| Elevation | 2.14 (0.04)[1] | 0.70 (0.79) |
| Slope % | 0.98 (0.55) | 1.03 (0.46) |
| Aspect | 1.25 (0.30) | 1.03 (0.46) |
| PDR | 0.86 (0.55) | 0.81 (0.62) |
| Slope shape | 1.35 (0.14) | 1.28 (0.20) |
| Stand age | 1.54 (0.16) | 1.46 (0.20) |
| *Sapling:* | | |
| Eigenvalue | 0.28 | 0.18 |
| Species-environment correlation | 0.80 | 0.66 |
| Cumulative % variance of species | 9.90 | 16.10 |
| Cumulative % variance of species-environment relation | 38.70 | 63.00 |
| Monte Carlo Permutation tests: | | |
| Elevation | 1.52 (0.11) | 1.43 (0.16) |
| Slope % | 1.47 (0.15) | 1.45 (0.14) |
| Aspect | 2.46 (0.03) | 2.41 (0.03) |
| PDR | 1.91 (0.02) | 0.43 (0.96) |
| Slope shape | 2.34 (0.03) | 2.45 (0.03) |
| Stand age | 2.03 (0.02) | 1.15 (0.33) |
| *Subcanopy:* | | |
| Eigenvalue | 0.53 | 0.29 |
| Species-environment correlation | 0.83 | 0.76 |
| Cumulative % variance of species | 15.10 | 23.40 |
| Cumulative % variance of species-environment relation | 51.0 | 79.10 |
| Monte Carlo Permutation tests: | | |
| Elevation | 0.38 (0.93) | 0.30 (0.96) |
| Slope % | 0.55 (0.76) | 0.64 (0.73) |
| Aspect | 3.88 (0.01) | 4.20 (0.01) |
| PDR | 0.95 (0.51) | 0.82 (0.50) |
| Slope shape | 4.46 (0.01) | 4.03 (0.01) |
| Stand age | 1.81 (0.08) | 1.46 (0.18) |
| *Seedling:* | | |
| Eigenvalue | 0.29 | 0.14 |
| Species-environment correlation | 0.80 | 0.68 |
| Cumulative % variance of species | 11.10 | 16.50 |
| Cumulative % variance of species-stand factor relation | 44.70 | 66.70 |
| Monte Carlo Permutation tests: | | |
| Elevation | 1.81 (0.04) | 0.88 (0.55) |
| Slope % | 0.72 (0.74) | 0.37 (1.00) |
| Aspect | 2.24 (0.01) | 1.75 (0.06) |
| PDR | 0.34 (0.99) | 0.31 (1.00) |
| Slope shape | 2.49 (0.02) | 1.23 (0.21) |
| Stand age | 2.72 (0.01) | 1.71 (0.12) |

[1] F-statistic and associated *P*-value for Monte Carlo permutation tests of stand factors with canonical axes.

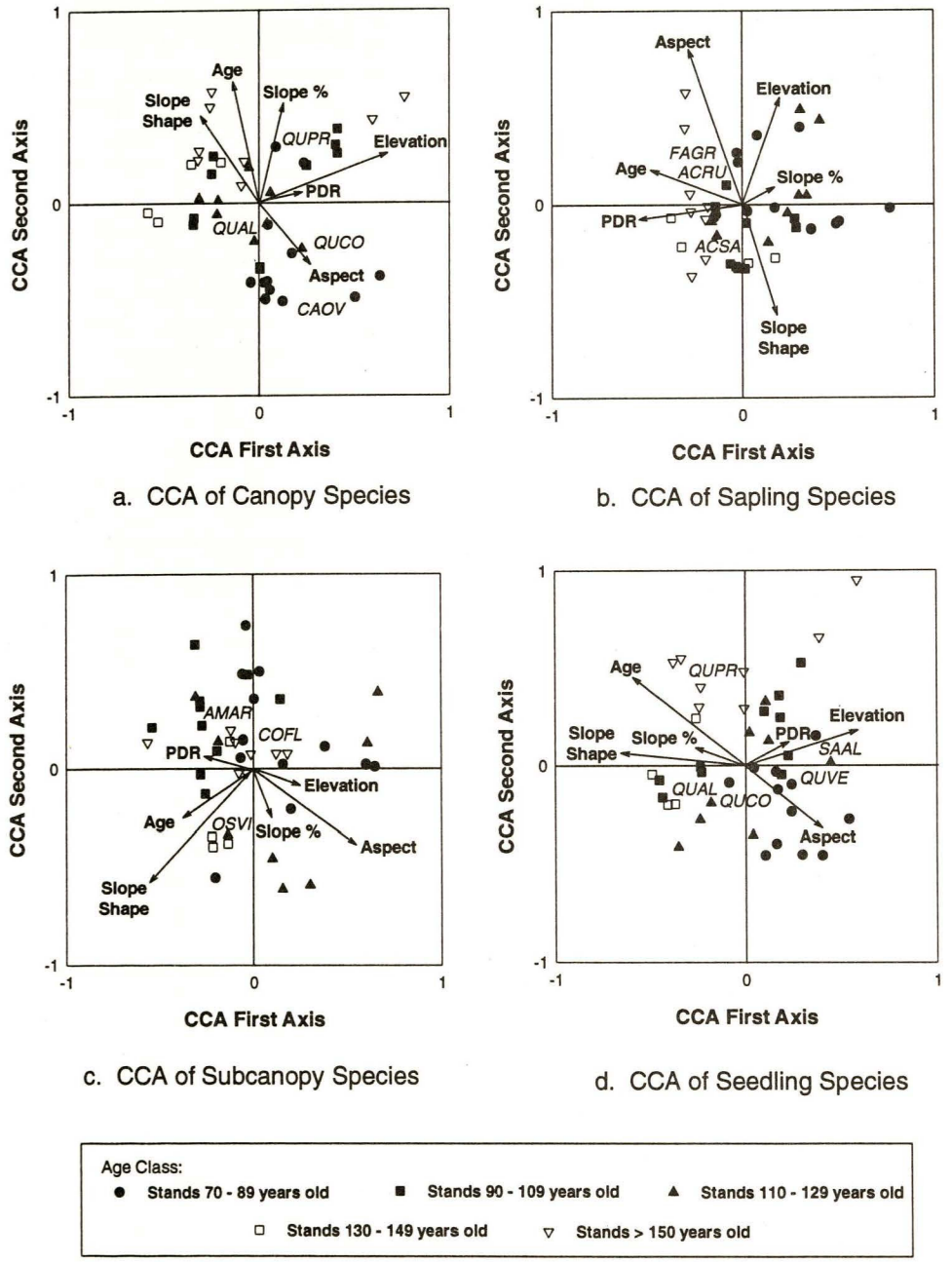


Figure 3. Canonical correspondence analyses (CCA) of canopy species (a), sapling species (b), subcanopy species (c), and seedling species (d) and stand factors.

Table 2. Importance values of selected canopy species, and relative densities of selected sapling, subcanopy, and seedling species by age class for second-growth and old-growth study sites in southeastern Ohio.

| Species Name | Acronym | Age Class (yrs) 70 - 89 | 90 - 109 | 110 - 129 | 130 - 149 | ≥ 150 |
|---|---|---|---|---|---|---|
| Number of plots (*n*) | | 12 | 10 | 8 | 4 | 8 |
| *Canopy Species:* | | | | | | |
| *Acer rubrum* | ACRU | 0.4 | 0.9 | 1.7 | 0.0 | 0.0 |
| *A. saccharum* | ACSA | 1.5 | 6.8 | 3.3 | 0.0 | 0.0 |
| *Carya glabra* | CAGL | 0.7 | 5.6 | 0.8 | 0.0 | 0.0 |
| *Quercus alba* | QUAL | 35.5 | 54.7 | 67.0 | 86.1 | 46.5 |
| *Q. coccinea* | QUCO | 11.8 | 2.7 | 1.0 | 0.0 | 1.6 |
| *Q. prinus* | QUPR | 12.2 | 12.2 | 2.8 | 0.0 | 27.6 |
| *Q. rubra* | QURU | 9.8 | 3.0 | 0.0 | 3.8 | 6.1 |
| *Q. velutina* | QUVE | 16.2 | 8.5 | 13.8 | 3.0 | 13.9 |
| Other[1] | | 11.9 | 5.6 | 9.6 | 7.1 | 4.3 |
| *Sapling Species:* | | | | | | |
| *Acer rubrum* | ACRU | 45.9 | 33.8 | 27.1 | 33.9 | 29.7 |
| *A. saccharum* | ACSA | 6.4 | 44.2 | 47.8 | 18.0 | 27.8 |
| *Fagus grandifolia* | FAGR | 9.4 | 6.4 | 14.7 | 18.2 | 28.3 |
| *Nyssa sylvatica* | NYSY | 1.3 | 0.0 | 0.8 | 8.6 | 4.9 |
| *Quercus alba* | QUAL | 3.1 | 0.0 | 2.1 | 0.0 | 1.1 |
| *Q. coccinea* | QUCO | 4.9 | 0.0 | 0.0 | 0.0 | 0.0 |
| *Q. prinus* | QUPR | 6.1 | 0.0 | 0.0 | 0.0 | 0.0 |
| *Q. rubra* | QURU | 1.9 | 0.6 | 0.0 | 0.0 | 0.0 |
| *Q. velutina* | QUVE | 0.8 | 0.6 | 2.3 | 0.0 | 0.0 |
| Other[2] | | 20.2 | 14.4 | 5.2 | 22.2 | 8.2 |
| *Subcanopy Species:* | | | | | | |
| *Amelanchier arborea* | AMAR | 10.0 | 7.5 | 33.0 | 13.0 | 13.0 |
| *Carpinus caroliniana* | CACA | 6.4 | 6.3 | 1.3 | 8.6 | 0.0 |
| *Cornus florida* | COFL | 63.3 | 60.0 | 16.6 | 31.3 | 72.7 |
| *Ostrya virginiana* | OSVI | 8.5 | 15.9 | 27.7 | 46.7 | 6.5 |
| Other[3] | | 11.8 | 10.3 | 21.4 | 0.4 | 7.8 |
| *Seedling Species:* | | | | | | |
| *Acer rubrum* | ACRU | 16.8 | 12.4 | 17.1 | 8.3 | 21.7 |
| *A. saccharum* | ACSA | 4.6 | 11.0 | 8.3 | 3.3 | 9.7 |
| *Fagus grandifolia* | FAGR | 3.8 | 3.5 | 3.3 | 3.3 | 9.0 |
| *Quercus alba* | QUAL | 4.9 | 22.7 | 10.9 | 49.0 | 7.0 |
| *Q. coccinea* | QUCO | 2.5 | 0.0 | 0.0 | 0.0 | 1.1 |
| *Q. prinus* | QUPR | 3.4 | 5.2 | 0.6 | 0.0 | 9.0 |
| *Q. rubra* | QURU | 2.5 | 1.5 | 1.5 | 1.8 | 1.0 |
| *Q. velutina* | QUVE | 3.2 | 2.3 | 2.5 | 0.0 | 1.3 |
| *Sassafras albidum* | SAAL | 16.3 | 12.3 | 13.0 | 4.0 | 12.8 |
| Other[4] | | 41.4 | 29.1 | 41.8 | 30.3 | 27.4 |

[1] *Carya cordiformis, C. lacinosa, C. ovata, C. tomentosa, Fraxinus pennsylvanica, Juglans nigra, Liriodendron tulipifera, Nyssa sylvatica,* and *Populus grandidentata.*

[2] *Aesculus octandra, Carya glabra, C. ovata, C. tomentosa, Diospryos virginiana, Fraxinus americana, F. pennsylvanica, Liriodendron tulipifera, Magnolia acuminata, Oxydendrum arboreum, Prunus serotina, Sassafras albidum,* and *Ulmus rubra.*

[3] *Asimina triloba, Cercis canadensis, Crataegus* spp., *Hamamelis virginiana, Lindera benzoin, Viburnum acerifolium,* and *V. prunifolium.*

[4] *Aesculus octandra, C. cordiformis, C. glabra, C. ovata, C. tomentosa, Cercis canadensis, Fraxinus americana, F. pennsylvanica, Liriodendron tulipifera, Magnolia acuminata, Nyssa sylvatica, Oxydendrum arboreum, Pinus strobus, Prunus serotina, Tilia americana,* and *Ulmus rubra.*

Finally, the seedling CCA revealed similar trends as the sapling CCA, with the stand factors explaining 66.7% of the variation in seedling composition (Table 1). In general, the seedling layers of the younger second-growth stands were comprised of scarlet oak, black oak, and sassafras (*Sassafras albidum* (Nutt.)), while those of the older second-growth stands were comprised of primarily of white oak (Table 2). The CCA triplot confirmed this trend, with stand age significantly, negatively associated with the first CCA axis ($P \leq 0.01$). Aspect was significantly and positively correlated with the first canonical axis, suggesting that the younger, more southerly facing slopes were associated with higher relative densities of scarlet oak and black oak seedlings ($P \leq 0.01$).

Canonical Correlations Among Forest Strata

The canonical correlation coefficients of the CCA axis scores varied considerably among the strata comparisons (Figure 4). The highest canonical correlations were between the canopy and seedling strata for both the first and second canonical axes ($P < 0.01$; $r = 0.63$ and 0.53, respectively), while the canonical correlations between the sapling strata and the seedling strata were also significant for both CCA axes ($P < 0.05$; $r = 0.50$ and $r = 0.30$, respectively). However, the canopy and sapling strata were not significantly correlated. Similarly, the subcanopy strata CCA axis scores were not significantly correlated with any of the other strata axis scores (Figure 4).

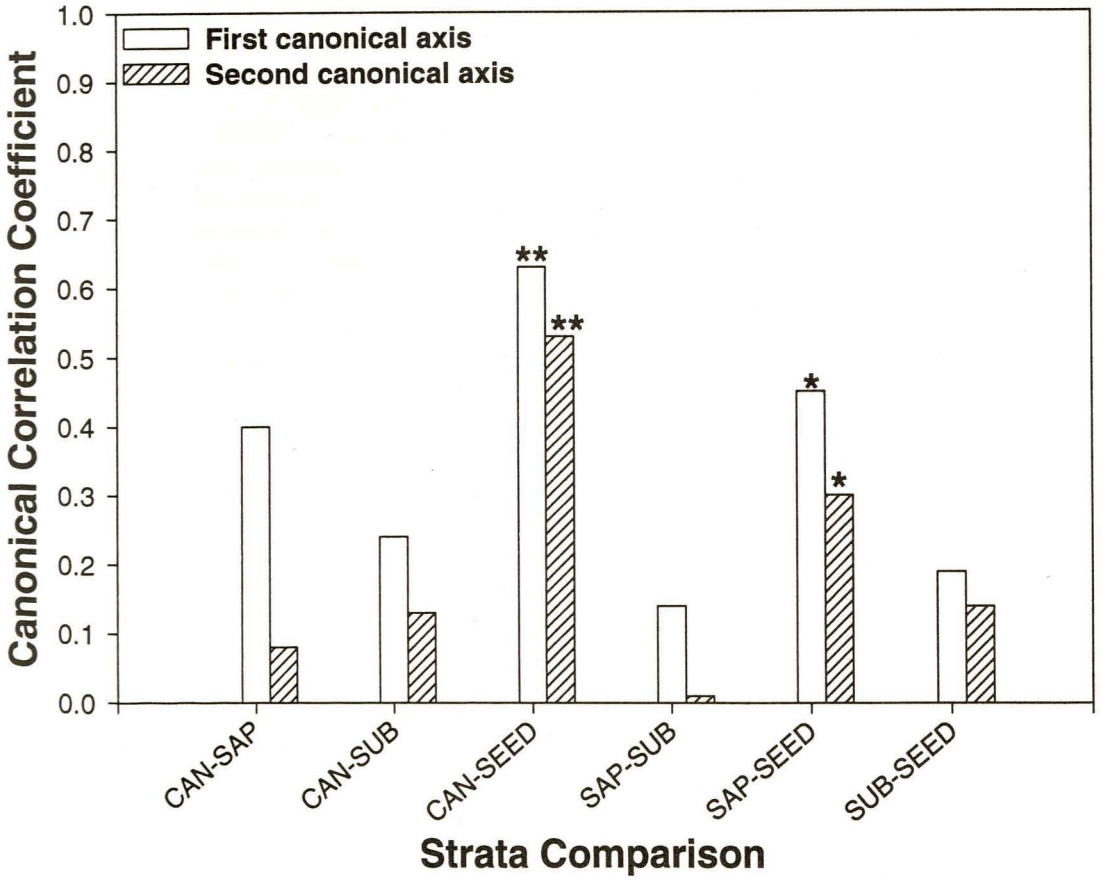


Figure 4. First and second axis canonical correlation coefficients of strata comparisons for second-growth and old-growth study areas in southeastern Ohio. CAN = canopy strata, SAP = sapling strata, SUB = subcanopy strata, and SEED = seedling strata. * = $P < 0.05$; ** = $P < 0.01$.

More detailed analysis of the significant canonical correlations revealed several interesting trends. The first and second axis scores of the canopy CCA and seedling CCA comparison were positively correlated with the their respective canonical variable (Table 3). Thus, the first and second axes of the canopy and seedling ordinations presumably represent the same gradients. As the canopy strata increased in white oak IV, the sapling strata also increased in white oak relative density. Because stand age is significantly correlated with the first CCA axis of the seedling ordination, the canonical correlation may represent the relationship between stand age and increasing importance of white oak in the canopy and seedling strata. The fact that the canopy and seedling strata are correlated is not surprising in that the correlation may reflect the strong influence of canopy trees on the composition of the seedling strata (e.g., local seed sources).

Similarly, the sapling strata and seedling strata were also significantly correlated (Figure 4). However, the relationships between the CCA axes were different than the canopy and seedling comparison. The first axis of the sapling ordination was weakly and negatively correlated with the first canonical variable, while strongly and negatively correlated with the second canonical variable (Table 4). Conversely, the seeding ordination was highly, positively correlated with the first and second canonical variables (Table 4). As a result, the second axis of the sapling CCA and the inverse of CCA axis 2 of the seedling ordination represent similar gradients. However, the variance extracted and redundancy values for both canonical variables are low, therefore caution should be exercised when interpreting these axes (Table 4).

## DISCUSSION

Several studies that have shown that there are strong relationships among vegetational strata in response to environmental and disturbance gradients (Grigal and Arneman 1970, del Moral and Watson 1978, Bradfield and Scagel 1984, Gagnon and Bradfield 1986, Roberts and Christensen 1988). Our data suggests a similar trend. Canonical correlation analyses indicate that the canopy and seedling strata, as well as the sapling and seedling strata, are correlated and are responding to similar gradients. However, the results of this study also show that compositional changes are not strongly related to stand age in all cases, although there do appear to be some consistent trends among the various strata. First, the younger second-growth stands 70 - 119 years old are comprised of mixed-oak canopies, red maple saplings, and mixed-oak seedlings. However, the composition of all three strata are markedly different in the older second-growth stands. The older second-growth stands 110 - 149 years old are

Table 3. Canonical correlation comparison of canopy strata with seedling strata ordination (CCA) scores.

| | CAN 1[1] | CAN 2 |
|---|---|---|
| Correlation of Canopy CCA Scores with their canonical variable | | |
| CCA Axis 1 | 0.94 | 0.33 |
| CCA Axis 2 | -0.45 | 0.89 |
| Variance extracted (%)[2] | 52.0 | 48.0 |
| Redundancy (%)[3] | 21.0 | 13.0 |
| Correlation of Seedling CCA Scores with their canonical variable | | |
| CCA Axis 1 | 0.99 | 0.11 |
| CCA Axis 2 | -0.01 | 0.99 |
| Variance extracted (%)[2] | 51.0 | 49.0 |
| Redundancy (%)[3] | 17.0 | 17.0 |

[1] Canonical variable.
[2] Proportion of the variance in the stand factors accounted for by the canonical variable.
[3] Proportion of variance in a vegetative stratum (as summarized by CCA axes 1 and 2) accounted for by the canonical variable.

Table 4. Canonical correlation comparison of sapling strata with seedling strata ordination (CCA) scores.

| | CAN 1[1] | CAN 2 |
|---|---|---|
| Correlation of Sapling CCA Scores with their canonical variable | | |
| CCA Axis 1 | -0.27 | 0.96 |
| CCA Axis 2 | 0.88 | 0.46 |
| Variance extracted (%)[2] | 51.0 | 49.0 |
| Redundancy (%)[3] | 10.0 | 5.0 |
| Correlation of Seedling CCA Scores with their canonical variable | | |
| CCA Axis 1 | 0.85 | 0.52 |
| CCA Axis 2 | 0.61 | -0.79 |
| Variance extracted (%)[2] | 52.0 | 48.0 |
| Redundancy (%)[3] | 10.0 | 4.0 |

[1] Canonical variable.
[2] Proportion of the variance in the stand factors accounted for by the canonical variable.
[3] Proportion of variance in a vegetative stratum (as summarized by CCA axes 1 and 2) accounted for by thecanonical variable.

dominated by canopies of white oak, saplings of sugar maple, and seedlings of white oak and subcanopy species (e.g., eastern hophornbeam). Finally, the old-growth stands appear to be comprised of canopy layers of mixed-oaks, sapling layers dominated by both red maple and sugar maple, and seedling layers of mixed-oaks. Since all study areas were located on a single ecosystem type with similar environmental characteristics, the changes in the vegetative composition are probably the result of modifications in the disturbance regimes (e.g., fire, grazing) rather than changes in site conditions or stand age.

The multivariate analyses used to describe the relationships between strata along the chronosequence suggest that the mechanisms regulating the development of these mixed-oak forest ecosystems in southeastern Ohio have been modified. Prior to European settlement and the subsequent changes in the disturbance regimes of the study area (see Goebel and Hix 1996), the primary mechanism regulating stand dynamics was most likely a class 3 effect (Swanson and others 1988). On south-facing ecosystem types surface fires most likely damaged or killed many of the red and sugar maple saplings and seedlings (Beatley 1959, Rympa 1969, Runkle 1990, Abrams 1992), inhibiting the development of a shade tolerant understory. Similar development of more shade tolerant understories on xeric sites in response to landform effects on disturbance patterns have been observed in other regions (Ware and others 1993, Goebel and others 1996).

With the active fire suppression activities of the past 80 years, the mechanism most likely inhibiting the development of shade tolerant understory strata was removed from the landscape. This change resulted in the development of a dense layer of red maple and sugar maple sapling layer. In the absence of a class 3 effect, the mechanism currently regulating stand development on these south-facing slopes appears to be a class 1 effect (Swanson and others 1988). Because many of the correlations between forest strata were weak, these class 1 effects may influence the development of various strata at different rates, as proposed by McCune and Antos (1981). The lack of correspondence between the canopy and sapling layers (e.g., few oaks in the sapling strata) suggests that red maple and sugar maple have survived in the understory of the mixed-oak canopies, and have overtopped the oaks of the seedling layer. Harvesting activities also have been shown to further the development of a shade tolerant understory on dry-mesic sites (Abrams and Nowacki 1992). Once the primary factor inhibiting the development of these shade tolerant understories was removed from the system and these understories become dominated by shade tolerant species, any natural or anthropogenic canopy disturbance will likely shift the canopy composition towards more shade tolerant species.

Although relatively undisturbed since establishment, the mechanisms regulating stand development in the old-growth stands of this study have also been modified. The old-growth areas sampled have been forested since European settlement and appear to have been in compositional equilibrium (Goebel and Hix 1996), the old-growth forests are no longer in equilibrium (e.g., canopy species are replacing themselves). Human activities in the surrounding matrix of second-growth forests such as logging, agricultural conversion, mining, and urban development have modified the landscape disturbance regimes of these mixed-oak ecosystems, primarily the frequency and intensity of surface fires. Without these natural disturbances, both red and sugar maple dominate the sapling layer of the old-growth study areas and will most likely increase in canopy importance in the future. Similar trends have been documented throughout the eastern United States (Whitney and Somerlot 1985, Lorimer 1989, Abrams 1992, Abrams and Nowacki 1992, Goebel and Hix 1996). Although the old-growth forests in the study area meet many of the structural and functional definitions of old-growth forests (e.g., high species richness, uneven-aged structure, large canopy individuals, relatively undisturbed soils), they are not in steady state with regard to species composition and other ecosystem properties because they are still responding to modified disturbance regimes. As a result, these old-growth forests are probably best described as changing, or transitional old-growth (Runkle 1996).

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# INDICATORS OF NITRATE EXPORT FROM FORESTED WATERSHEDS OF THE CHESAPEAKE BAY REGION

Karl W.J. Williard[1]

Abstract: Soil net nitrogen mineralization and nitrification rates were studied on nine relatively undisturbed, forested watersheds in an effort to explain the large variations in nitrate export in streamflow within the Chesapeake Bay region. The primary hypothesis tested was that nitrate export from the watersheds was positively associated with rates of net soil nitrogen mineralization and net nitrification in the upper 10-cm of mineral soil. Rates of soil net nitrogen mineralization and net nitrification were measured over a 5-week summer incubation period (June-July) using nine buried bags in each of the three major soil types on each watershed. High, medium, and low nitrate export watersheds, respectively, exhibited the high, medium, and low mean net nitrogen mineralization and net nitrification rates at $\alpha = 0.05$. C/N ratios and soil moisture content together explained 44% of the variation in soil nitrogen mineralization rates and 41% of the variation in soil nitrification rates. The percent of total nitrogen in the upper 10-cm of mineral soil explained 46% variation in watershed nitrate export, while the percentage of mineralization due to nitrification explained 42% of the variation in watershed nitrate export. Estimated rates of wet and dry atmospheric deposition of nitrogen were not significantly correlated with watershed nitrate export. The percentage of N in the upper mineral soil and soil net mineralization and nitrification rates show promise as indicators of nitrate export from forested watersheds within the Chesapeake Bay region.

[1]USDA Water Science Fellow, Environmental Resources Research Institute, Pennsylvania State University, University Park, PA 16802.

# THE EFFECTS OF PARTIAL CUTTING PRACTICES ON FOREST STAND STRUCTURE IN APPALACHIAN HARDWOOD FORESTS

Mary Ann Fajvan[1] and Shawn T. Grushecky[2]

Abstract: Eastern hardwood forests originated after catastrophic disturbances around the turn of the century and are currently an even-aged, maturing resource. The increasing value of sawlogs, especially those of particular species and quality, has prompted many forest landowners to increase their harvesting efforts. Most harvesting appears to be economically driven, focusing on sawlog extraction without regard to the future of the residual stand. Partial cutting practices, such as diameter limit cutting, remove the highest quality material and leave inferior species and quality behind. None of these cutting practices are part of any planned silvicultural system and may alter future forest sustainability.

In 1993 a study was initiated on the West Virginia University Forest to compare diameter-limit harvesting practices with a silvicultural regeneration method: the shelterwood method. Harvested areas are 4 ha (10 a) and there are four replications of three treatments: (1) diameter-limit cutting where all merchantable trees > 30 cm (12") dbh were removed, (2) diameter-limit cutting where all merchantable trees > 40 cm (16") dbh were removed, (3) seed cut of the shelterwood method. Stands were 60 years old and harvesting occurred during the fall of 1993 and early spring 1994. Stand characteristics were measured before harvest and so far, for three growing seasons post-harvest.

During the winter of 1995, a 1 ha (2.47 a) block was established near the center of each treatment area. Within each block, the location of each residual stem was mapped and diameter, height, crown area and live crown length were measured using a survey laser. The objective of this part of the study is the long-term monitoring of residual tree growth. Results from the 1 ha mapped blocks (architectural maps) show that the treatments have resulted in very different stand structures. Residual basal area of the shelterwood cut was double (19 $m^2$/ha) that of the diameter limit cuts and 92% of the shelterwood basal area/crown area was in trees > 21 m (70') tall. Whereas in the diameter-limit harvests approximately 25% of the basal area was in trees between 12.3-21.3 m (41-70') tall and a higher percentage of crown area (35-45%) was found in this canopy layer.

Because the canopy mapping is such a time-consuming process, the canopy maps were used as baseline data for testing the use of geostatistical sampling techniques to predict canopy structure. A 6 m x 6 m grid was established within each 1 ha plot for a total of 289 points. At each point an observer looked straight up through a sighting tube to determine whether or not canopy was present; the observer also looked straight down to detemine if regeneration was present. Semivariograms and kriging were used to model and map the spatial pattern and structure of canopy. Kriged maps showed a higher percentage of the area consisted of "gaps" in the diameter-limit harvests compared to the shelterwood. Kriged maps were compared to architectural maps and the accuracy of predicting presence/ absence of canopy ranged between 54-56%. Seventy-five percent of the error came from incorrectly noting canopy presence in field and 25% came from incorrect gap predictions from the model. In the next phase of the project, the scope and methodology will be modified to minimize these types of errors.

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[1]Assistant Professor of Silviculture, Division of Forestry, West Virginia University, P.O. Box 6125, Morgantown, WV 26506-6125.

[2]Research Assistant, Division of Forestry, West Virginia University, P.O. Box 6125, Morgantown, WV 26506-6125.

# EFFECTS OF SEASONAL PRESCRIBED FIRES ON HARDWOOD ADVANCE REGENERATION IN SHELTERWOOD STANDS

Patrick Brose and David Van Lear[1]

Abstract: Shelterwood harvesting of mature oak (*Quercus* spp. L.) stands on productive sites often fails because fast-growing intolerant and already established tolerant species outcompete oak reproduction for dominance of the advance regeneration pool. We hypothesized that prescribe fire would improve the competitive position of oak in the advance regeneration pool of shelterwood stands because of different growth strategies among species.

Three mature mixed oak stands on productive sites were partially harvested using the shelterwood system. Two to 4 years later, each stand was divided into spring burn, summer burn, winter burn, and control treatments. All treatments were systematically inventoried for advance regeneration using permanent plots before and after burning. Variables measured were density, form, growth rate, height, and mortality rate of hardwood advance regeneration. Prescribe fires top-killed hardwood regeneration, forcing the stems to resprout. Fire intensity was monitored with heat-sensitive paints and by estimating woody fuel consumption.

Oak density was not reduced by any prescribe fire and produced straight vigorous new stems that showed accelerated growth for 2 years. Hickory (*Carya* spp. Nuttall) responded to burning similarly to oak. Yellow-poplar (*Liriodendron tulipifera* L.) density was reduced by all burning treatments and resprouting stems exhibited accelerated growth for only 1 year. Red maple (*Acer rubrum* L.) density was significantly decreased by moderate to high intensity growing-season fires and it also displayed accelerated growth for only 1 year. Overstory damage and mortality was minimal, restricted to thin-bark species and trees with slash at their bases. All prescribe fires improved the competitive position of oak reproduction in the advance regeneration pool with spring fires providing the best blend of competition control, oak stem form improvement, and growth acceleration. These results indicate that prescribe burning as a follow-up treatment to shelterwood harvesting is a promising approach to regenerating oak stands on productive sites.

[1] Graduate Research Assistant and Bowen Professor of Forestry, respectively, Department of Forest Resources, Clemson University, 261 Lehotsky Hall, Clemson, SC 29634.

# GRASS OR FERN COMPETITION REDUCE GROWTH AND SURVIVAL OF PLANTED TREE SEEDLINGS

Larry H. McCormick and Todd W. Bowersox[1]

Abstract: Bareroot seedlings of northern red oak, white ash, yellow-poplar and white pine were planted into herbaceous communities at three forested sites in central Pennsylvania that were clearcut 0 to 1 year earlier. Seedlings were grown 4 years in the presence and absence of either an established grass or hay-scented fern community. Survival and height growth were measured annually at the end of each growing season. Fourth year results showed significant decreases in the average survival of northern red oak, white ash and yellow-poplar seedlings grown in the presence of grass or hay-scented fern. Survival of the white pine was not affected by the presence or absence of either herbaceous community. Tree heights were also affected by herbaceous competition. After four growing seasons, the average heights of all tree species were significantly greater in areas maintained free of grasses or hay-scented fern. Average height responses due to release from herbaceous competition were greatest for white ash and yellow-poplar with average increases of up to 150% after four growing seasons. White pine showed the least response to control of herbaceous vegetation with average height increases of < 30%. The results of this study demonstrate the importance of controlling herbaceous vegetation when artificially regenerating hardwoods on forested sites that have been clearcut.

## INTRODUCTION

A recent statewide inventory of Pennsylvania's forests, has raised concerns about the likelihood of these forests to successfully regenerate following a natural disturbance or harvest cut (Williams and others 1995). The inventory revealed that a large majority of the sampled forest stands with 40% to 75% overstory stocking did not have adequate amounts of advance tree-seedling regeneration for new stand establishment. Such findings emphasize the continuing need to develop effective means of regenerating hardwood stands.

Past efforts to obtain successful regeneration of harvested Allegheny hardwood and mixed oaks stands in Pennsylvania have focused on enhancing natural regeneration. Use of electric fences to reduce deer browsing, chemical control of competing woody and herbaceous understory vegetation, partial overstory removals prior to the final harvest cut, and fertilization following harvesting are some of the methods that have been explored. While increasing the likelihood of obtaining successful regeneration, these measures have not consistently provided sufficient numbers of seedlings to ensure adequate distribution of seedling stocking or desirable species compositions in all stands. This suggests the possible need for artifical regeneration.

Studies by Johnson (1976, 1981, 1984) with northern red oak and Francis and Bivens (1985) with yellow-poplar and black cherry have demonstrated the potential for hardwood plantings in forested stands. Unfortunately, previous plantings of hardwoods have not routinely resulted in acceptable survival and growth when compared to naturally regenerated hardwoods or conifers. The "failure" of these plantings has frequently been attributed to stresses from excessive undesireable vegetation (Sander 1971, Russell 1984, von Althen 1977, Loftis 1979, Johnson 1981, McGee 1981). Competition from herbaceous vegetation has traditionally been a problem in old field plantings of hardwoods. The same may be true for forest plantings as many of the harvested sites in the Northeast are rapidly invaded by persistent herbaceous vegetation (Williams and others 1995).

To assess the potential importance of this problem, a study was conducted to examine the effect of herbaceous vegetation on the survival and growth of barerooted seedlings planted in oak stands of Pennsylvania. Four species,

[1]Professors, School of Forest Resources, The Pennsylvania State University, University Park, PA 16802.

northern red oak (*Quercus rubra* L.), white ash (*Fraxinus americana* L.), yellow-poplar (*Liriodendron tulipifera* L.), and white pine (*Pinus strobus* L.) were grown in the presence and absence of two herbaceous communities, hay-scented fern (*Dennstaedtia punctilobula* L.), and poverty oat grass (*Danthonia spicata* (L.) Beauv.). Results of the study after two growing seasons were reported earlier (Bowersox and McCormick 1987). The purpose of this paper is to provide an update of the results after four growing seasons.

## METHODS

The study consisted of two experiments, one initiated in the spring of 1984 and the other in the spring of 1985. Both experiments were conducted at two forest sites, Laurel Run and Sand Knob, located in central Pennsylvania near State College, PA, on lands managed by the Pennsylvania Bureau of Forestry. The sites previously contained 100-year-old, even-aged oak stands that were partially harvested in the early 1970s and then clearcut and fenced to reduce deer browsing in the 1983-1984 dormant season. A five strand electric fence was used to minimize deer browsing. Effectiveness of the fence decreased after 2 years of operation (George and others 1991). Following the partial cutting in the early 1970s, the sites became dominated by grass and fern communities. The grass communities were primarily poverty grass, but also contained lesser proportions of *Rubus* L., goldenrod (*Solidago* L.), fireweed (*Erechitites hieracifolia* (L.) Raf.), whorled loosestrife (*Lysimachia quadrifolia* (L.) Ell.), and sheep sorrell (*Rumex acetosella* L.). The fern communities were dominated by hay-scented fern. Soils at the sites had similar surface texture and chemical properties (extremely acidic—pH 4.6—and low fertility), but differed in drainage. The soil at Laurel Run was a poorly drained Andover loam (Typic Fragiaquults) whereas the soil at Sand Knob was a well-drained Hazelton sandy loam (Typic Dystrochrepts).

The experiments used a split-plot, randomized complete block design with four replications per site in 1984 and 1985. The main plots (160 $m^2$) initially contained either grass or fern communities. Each main plot was divided equally with one-half the area retained in either grass or ferns and the other one-half treated with glyphosate (1.7 kg ai/ha) to create grass-free or fern-free conditions. The glyphosate was applied in August 1983 (prior to harvest) for the 1984 planting and in August 1984 (first growing season after harvest) for the 1985 planting. In each weed and weed-free subplot, 20 bareroot seedlings each of northern red oak (1-0), white ash (1-0), yellow-poplar (1-0), and white pine (2-0) were planted at a 1x1 m spacing (2 rows of 10 trees) in early May. All seedlings were from seeds collected in Pennsylvania and grown in the state operated nursery. All hardwood seedling shoots were top-clipped to a 10 cm length prior to planting. The roots were pruned to fit into a 10 cm wide and 25 cm deep hole created by a KBC planting bar. The white pine seedlings averaged about 10 cm in stem length. Over all treatment combinations and replications, a total of 640 seedlings per species were planted in 1984 and in 1985. Annually in June-July the Grass-free and Fern-free treatment areas were hand-weeded around individual seedlings and the remaining vegetation controlled by spraying with glyphosate (1.7 kg ai/ha) to maintain weed free conditions.

Survival and total height values were measured at the end of the growing season for 4 years in 1984 and 1985 plantings. Species specific data were analyzed by analysis of variance with year, location, replication, herbaceous community as main effect factors and herbaceous conditions as a split-plot factor. All treatment effects were tested at the $p < 0.05$ level. Mean separations were performed using Duncan's Multiple Range Test. Only data from unbrowsed (deer) seedlings were used in the height growth analyses. This resulted in sample sizes for year four in the combined 1984 and 1985 height analyses of 380, 757, 568, and 676 for northern red oak, white ash, yellow-poplar, and white pine, respectively.

## RESULTS AND DISCUSSION

### Survival

Site was not a significant factor affecting survival, however, there were significant differences in survival associated with year of planting and treatment (Fern, Grass, Fern-free, and Grass-free). In general, survival was equally good for all Grass-free and Fern-free treatments in both the 1984 and 1985 plantings. Average survival 4 years after planting ranged from a low of 76% for 1985 planted yellow-poplar to 98% for 1984 and 1985 planted ash (Figure 1).

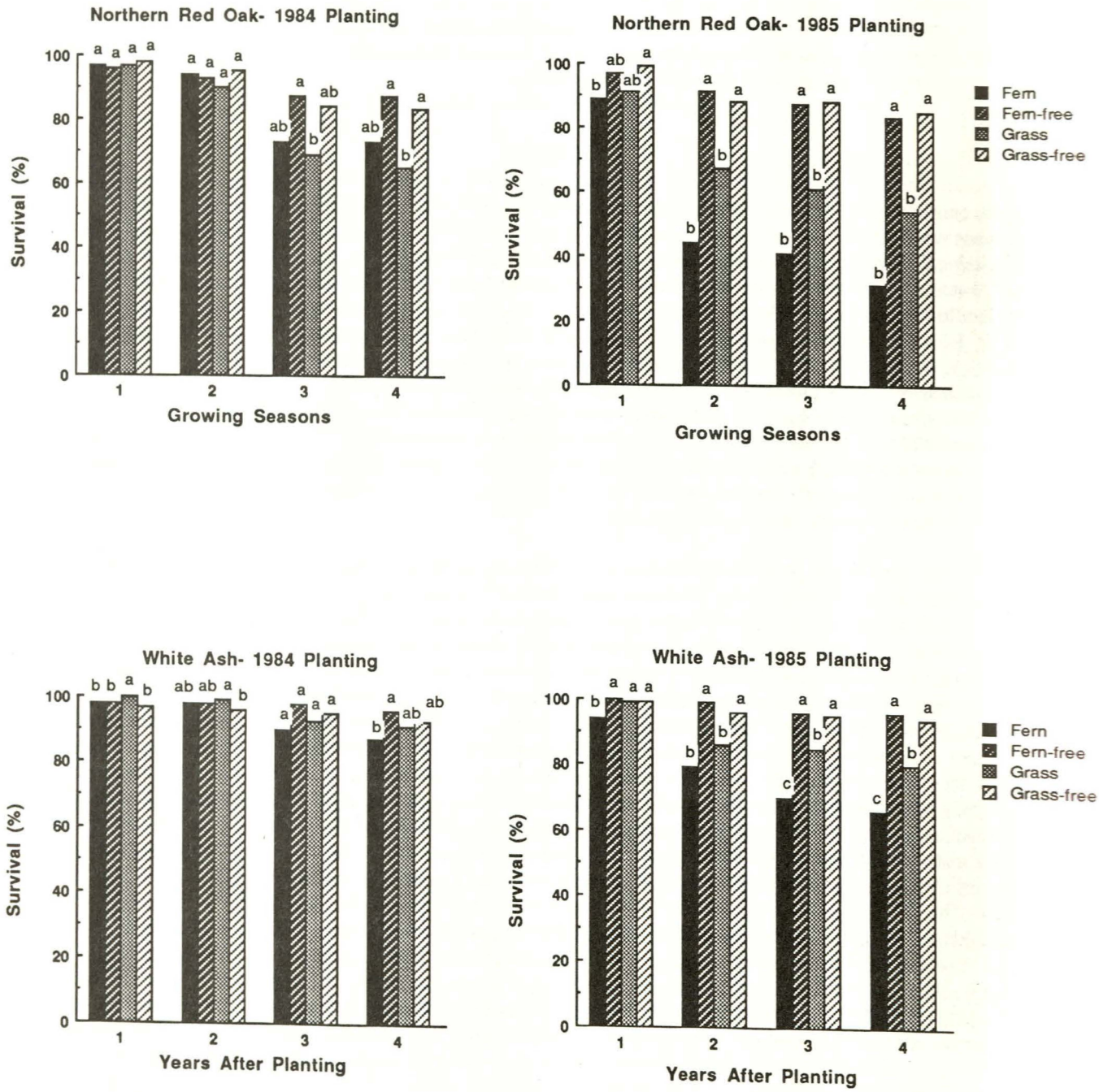


Figure 1. Average survival for 1 through 4 years after planting for the Fern, Fern-free, Grass, and Grass-free treatments, by planting years and species. Treatment survival averages (N=160) with the same letter within year after planting, planting year, and species were not significantly different at $p<0.05$.

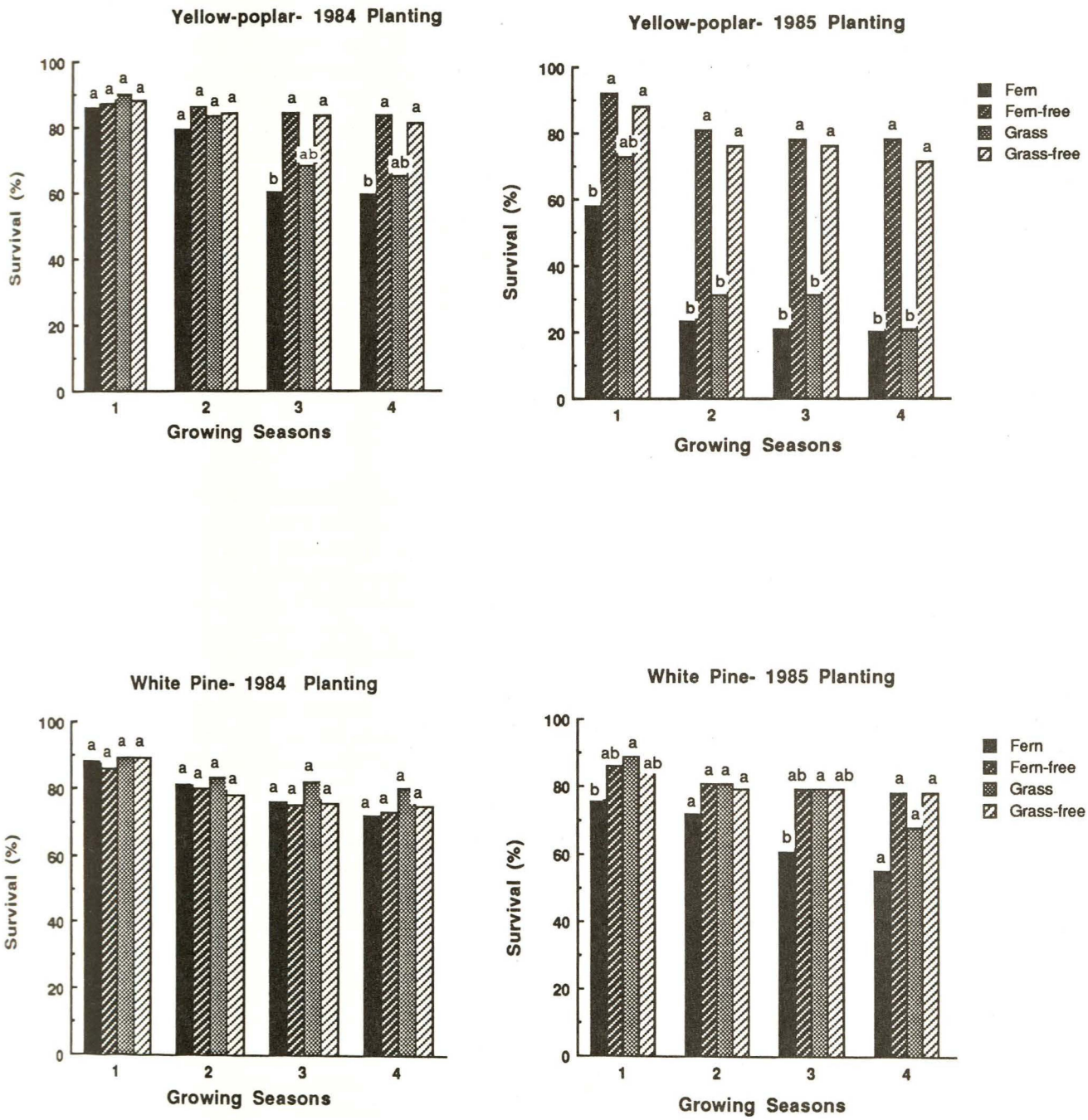


Figure 1 (cont.). Average survival for 1 through 4 years after planting for the Fern, Fern-free, Grass, and Grass-free treatments, by planting years and species. Treatment survival averages (N=160) with the same letter within year after planting, planting year, and species were not significantly different at $p<0.05$.

In contrast, seedling survival under weedy conditions tended to be lower—especially for the hardwoods. In the 1985 plantings, all of the hardwood species had significantly lower survival rates in the Grass and Fern treatments than in the Grass-free and Fern-free treatments after four growing seasons. Four year survival rates for the hardwoods in the 1985 plantings weedy treatments ranged from a low of about 20% for yellow-poplar (both Grass and Fern) to a high of about 70% for white ash (Grass). Survival of the white pine was not significantly affected by treatment. Similar response occurred in the 1984 plantings although hardwood survival rates of the in the Grass-free and Fern-free treatments were on average higher than in the 1985 planting. Rates of survival for the hardwoods tended to be lower in Fern treatments than in the Grass treatment, however, with the exception of white ash survival in 1985, the differences were not significantly different.

In both the 1984 and 1985 plantings, most of the seedling mortality in the Grass or Fern treatments occurred in the same calendar year (1986). This corresponded to the third year after planting for the 1984 planting and the second year after planting for the 1985 growing season. We are not certain what causes the increased level of mortality in 1986. Initially it was thought that this mortality may have been related to moisture stress. A check of precipitation data for 1985 and 1986 growing season, however, showed normal seasonal levels of precipitation. Other factors that could have accounted for the mortality include small mammal damage, deer browsing, or accumulated stresses associated with increased levels of competition from a maturing grass or fern communities.

Height Growth

Neither site nor year of planting significantly affected height growth of the planted seedlings. However, significant differences in height growth did occur among treatments. Starting in the second year after planting and continuing through the fourth year after planting, the average heights of all species were significantly greater in the Fern-free or Grass-free treatments than in the Fern or Grass treatments (Figure 2). The relative magnitude of the height differences between the weed and weed free treatments varied among species. White ash and yellow-poplar showed the greatest response to release from herbaceous competition. After four growing seasons, the average height of yellow-poplar trees in the Fern-free treatment was 1.9 times greater than in the Fern treatment and 1.6 times greater in the Grass-free than in the Grass treatment. Similarily, 4 years after planting, average height of white ash trees were 2.6 and 1.8 times greater in the Fern-free and Grass-free treatments, respectively, than their weedy counterparts. Height differences between the weed and weed free treatments were smaller for northern red oak and white pine. Average heights of the northern red oak and white pine in the Grass-free treatment after 4 years were only about 1.2 times greater than in the Grass treatment.

Type of herbaceous vegetation also had an effect on height growth of some species. After 4 years, the average heights of white ash and yellow-poplar in the Fern treatment were significantly different from the Grass treatment. These differences began to appear as early as 2 years after planting, and by the fourth year the average heights of white ash and yellow-poplar in the Fern treatments were about 20% less than the Grass treatments.. The average heights of the northern red oak and white pine were not significantly different between the Fern and Grass treatments. In contrast, the average height of northern red oak, white ash and white pine in the Fern-free treatment were greater than in the Grass-free treatment.

Results of this study indicate that planting can be a viable option for establishing northern red oak, white ash, yellow-poplar and white pine seedlings in clearcuts within the mixed oak forests of Pennsylvania. Overall, the fourth year survival and height values were acceptable for 1-0 bareroot northern red oak, white ash and yellow-poplar and 2-0 white pine seedlings. Average survival across all treatments was lowest for yellow-poplar (60%), moderate for northern red oak (70%) and white pine (72%), and highest for white ash (88%). Average height across all treatments was 1.1, 1.2, 2.0 and 3.0 m for white pine, northern red oak, white ash and yellow-poplar, respectively. Although these survival and height values may be acceptable, this study clearly indicates that the presence of herbaceous plants typical of disturbed mixed oak stands in Pennsylvania can have a detrimental effect on survival and growth of planted seedlings. This was especially true for hardwood species like northern red oak, white ash and yellow-poplar whose fourth year survival and height values were reduced when planted in either hay-scented fern or grass communities.

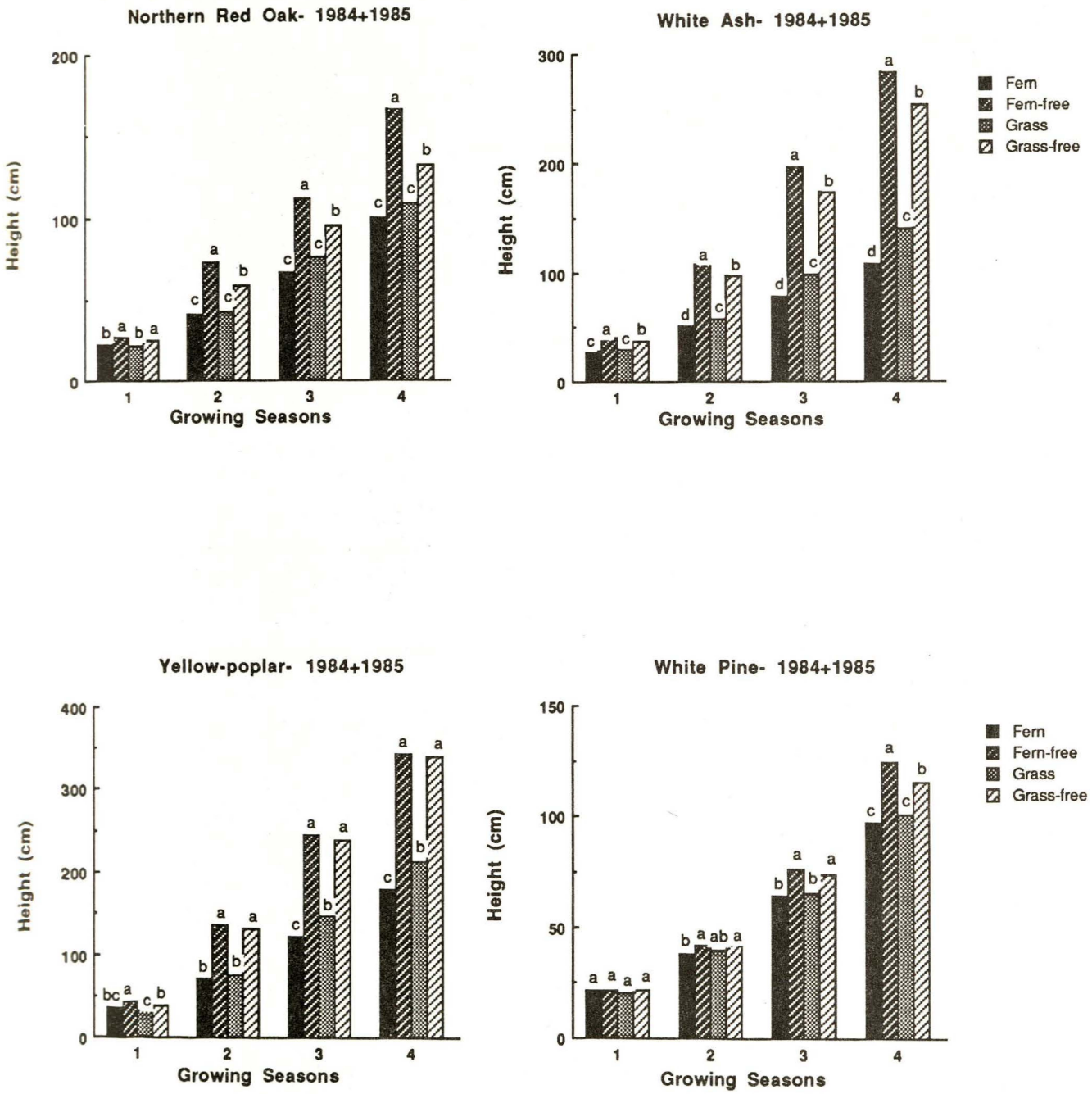


Figure 2. Average height for 1 through 4 years after planting for Fern, Fern-free, Grass, and Grass-free treatments by species. Combined 1984 and 1985 planting treatment values within a species and years after planting with the same letter were not significaty different at $p < 0.05$. Sample size was 380, 757, 568, and 676 for northern red oak, white ash, yellow-poplar, and white pine respectively.

Overall, northern red oak seedlings growing in the presence of either hay-scented fern or grass communities had a fourth year average survival of 56% and height of 1.1 m. When these herbaceous plants were treated with glyphosate and maintained weed-free, northern red oak fourth year survival averaged 85% and the height averaged 1.5 m. White ash fourth year survival was very good in both the weed-free (95%) and weed (81%) conditions but height was substantially different between the two herbaceous conditions. Growing in the presence of fern or grass, the white ash averaged 1.2 m in height after four growing seasons; whereas in the freedom of these herbaceous plants, white ash averaged 2.7 m in height. Yellow-poplar had the greatest responses to the control of ferns and grasses. In the presence of the two weedy conditions, yellow-poplar had an average survival of 41% and height of 2.0 m 4 years after planting. In the absence of these herbaceous plants, the 4 years after planting survival averaged 78% and height averaged 3.4 m. White pine was the least responsive to the control of the two herbaceous plants. Compared to the weedy condition, the weed-free treatments increased the average survival from 68% to 76% and height increased from 1.0 m to 1.2 m, respectively, 4 years after planting.

We believe that this study demonstrates that northern red oak, white ash, yellow-poplar, and white pine can be successfully planted into mixed oak stands of Pennsylvania. However, for the planted seedlings to have desirable survival and height growth in clearcut site dominated by herbaceous vegetation, some method of vegetation control must be used.

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# INTERFERENCE WITH SHOOT GROWTH AND FLOWERING OF DITTANY (*CUNILA ORIGANOIDES* (L.) BRITTON) BY HARDWOOD LEACHATES

J. W. Van Sambeek[1], Jane M. Kobe[2], and James S. Fralish[2]

Abstract: This study examined the potential role of hardwood leachates in the reduction of understory forbs in oak-hickory stands following invasion by sugar maple and dense-canopy shrubs. We conducted a shaded greenhouse experiment during two summers using stonemint dittany (*Cunila origanoides* (L.) Britton) as an indicator species. We used a modified stair-step bioassay with donor pots producing two levels of recycled leachates from hardwood seedlings root exudates, foliage washes, and decaying leaves. At the lower density, each donor pot had three hardwood seedlings and three applications 300 $cm^2$ of mature leaves of the same species to the litter layer (LAI = 1.5). At the higher density, we doubled the number of seedlings and surface area of mature leaves in the litter layer (LAI=3.0). Donor species tested for 1 or 2 years included sand-only, sugar maple, white oak, flowering dogwood, pawpaw, and black walnut. Dittany stem height ranged from 38 cm when exposed to leachates from white oak seedlings to 49 cm when exposed to leachates from sand-only donor pots. Dittany stem dry weights showed no differences in either year in response to leachates produced by different hardwood seedlings or at the two donor densities. Most dittany stems produced nearly 2,000 flowers during a 10 week period which began with completion of axillary shoot elongation. No differences in date of flower initiation or cumulative flower production were found in response to any of the hardwood leachates. We conclude that allelopathic interference of dittany by hardwoods, if it occurs at all, may require more stressful conditions than were created in this study and that the absence of dittany in oak-hickory stands after the invasion by sugar maple or pawpaw results primarily from competition for light and other resources.

## INTRODUCTION

The oak-hickory forests in the central hardwoods region were heavily disturbed by fire before being cut during post-European settlement (Fralish 1991). Without periodic fire, the regenerated oak forests are now being replaced by sugar maple and dense-canopy shrubs such as pawpaw on the more mesic sites and by flowering dogwood on the drier sites (Parker and others 1985, Schlesinger 1989). As the mid-canopy layer increases in density, this succession typically is accompanied with a loss of forest forbs such as stonemint dittany (*Cunila origanoides* (L.)Britton).

Although little information has been published on dittany, personal observations confirm dittany is a common aromatic forb found in most mature oak-hickory forests of the central hardwoods region. In the spring, multiple stems emerge from underground perennial storage roots. During the summer, the semi-decumbent woody stems reach lengths of 30 to 60 cm before initiating rapid axillary shoot elongation. Insect-pollinated flowers are formed in clusters along the primary and axillary shoots in early to late fall. We found no information on the mechanisms through which hardwood overstories might affect dittany growth and reproduction as overstory densities increase.

Plant-plant interference can result either from competition for resources or allelopathy (Horsley 1991, Rice 1987). Competition involves removal of essential resources by adjacent plants while allelopathy involves the production and release of chemicals that alter the competitive ability of other plants. The donor plant may affect the germination or growth of the receiver plant through release of allelochemicals from the foliage or roots, through microbial formation of allelochemicals within the rhizosphere, or by affecting bacteria and mycorrhizae populations important for cycling soil nutrients, especially nitrogen and phosphorus (Horsley 1993).

[1] Research Plant Physiologist, USDA Forest Service, North Central Forest Experiment Station, 1-26 Agriculture Building, University of Missouri, Columbia, MO 65211-0001.

[2] Former Graduate Assistant and Associate Professor Emeritus, respectively, Department of Forestry, Mailcode 4411, Southern Illinois University, Carbondale, IL 62901-4411.

Well-documented cases exist describing how grasses or forbs interfere with the regeneration of hardwoods (Fisher 1980); however, few studies document that the production of allelochemicals by hardwoods gaves them a competitive advantage over the competing understory. Rietveld (1983) showed that juglone, an allelochemical produced by black walnut, could inhibit germination and growth of many grasses, forbs, shrubs, and trees. Similarly, Gant and Clebsch (1975) suggested that sassafras produces allelochemicals that effectively eliminated 10 of the 37 forest forbs surveyed.

Separating the effects of allelopathy from those of competition, herbivory, disease, and other stresses under natural conditions is extremely difficult (Putnam and Tang 1986, Horsley 1991). The most common method for showing putative allelopathy is with seed germination and seedling growth bioassays, although novel approaches using in vitro bioassays have been tried (Preece and others 1991). Unfortunately, these bioassays frequently use concentrated plant extracts or leachates that may not occur in natural systems. In addition, most germination and growth bioassays used to date eliminate the associated microorganisms found in natural ecosystems.

The "stairstep bioassay" was designed to incorporate more of the components found in natural systems while still separating the effects of allelochemicals from competition for light, water, and nutrients (Lovett and Jokinen 1984, Quasem and Hill 1989). With the stairstep bioassy, allelochemical concentrations are allowed to increase slowly over time as the leachates are continuously recycled between the receiver plants and donor system including test plants, decaying plant parts, and developing litter layers. Various modifications such as addition of soil cores allow for normal destruction or conversion of suspected allelochemicals (Horsley 1991). Other modifications have been made to evaluate the possible interactions between allelochemicals and moisture stress (Rink and Van Sambeek 1985, 1987), nutrient stress (Stowe 1979), or various microorganisms (Qasem and Hill 1989).

Our objective was to learn whether growth and reproduction of dittany were affected by allelochemicals produced by exudates and decomposing leaves of white oak (*Quercus alba* L.), of dense canopy species like sugar maple (*Acer saccharum* Marsh.), pawpaw (*Asimina triloba* (L.) Dunal), flowering dogwood (*Cornus florida* L.), or of black walnut (*Juglans nigra* L.) which is known to produce the toxic allelochemical juglone. Experiments were designed to test two levels of leachate formation under conditions that included microorganisms associated with dittany grown under field conditions.

## METHODS

Our greenhouse study was designed as two dose-response experiments in which dittany plants were grown in sand culture and watered by leachates from four test hardwoods and a sand-only control. For this study, three coats of calcium carbonate were applied to the greenhouse ceiling each spring to reduce light intensity by 92% to 140 $umol \cdot m^{-2} \cdot s^{-1}$ on clear summer days. Four greenhouse benches were separated to create south-facing stairsteps 60 and 120 cm above the concrete and gravel floor as shown in Figure 1. Each bench served as one complete block with 10 experimental units in a 5 (donor species) x 2 (donor density) factorial arrangement of donor treatments.

Dittany plants with stems approximately 10 cm long were collected in mid-April 1991 from a local oak-hickory stand. Each plant was dug keeping 0.2- to 1.5-kg of water-saturated soil around the root system. Two plants were planted in each of forty 20 L plastic pots (30 cm diameter by 40 cm deep). Before planting, the soil from plants with a large soil and small soil mass were pressed together and placed on a layer of washed river sand in the bottom of each pot. Holes were punched in the top of the soil to direct movement of leachates through the soil before the pot was filled with sand. Each pot was placed on a 30 x 45 cm fiberglass tray. A total of 10 pots were placed along the lower level of each bench. In 1991, each pot of dittany was thinned to a single stem from each root ball. After the dittany shoots were harvested in 1991, the pots containing the dittany plants were labelled and moved to a large cooler at 4°C until April 1992. The pots were returned to their original position in April 1992. After shoot emergence, each pot was thinned to three stems (one in the center and one each on opposite sides of the pot) for the 1992 experiment.

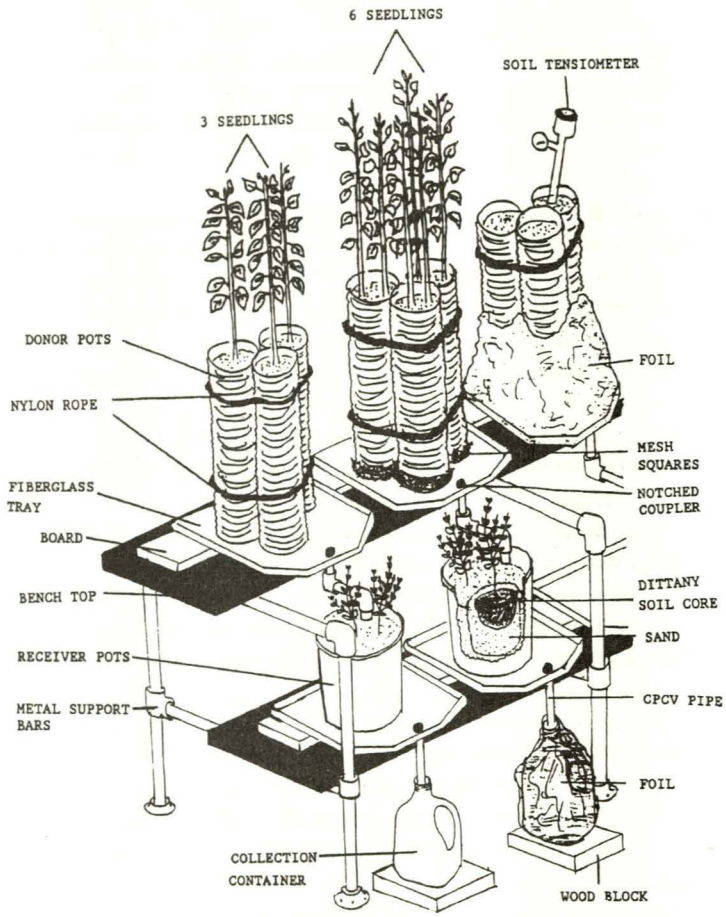


Figure 1. Diagram showing how experimental units were assembled.

Hardwood seedlings (1-0 stock) of sugar maple, white oak, flowering dogwood, and pawpaw were obtained from the Illinois state tree nursery or a commercial tree nursery. In mid April 1991, one or two hardwood seedlings were planted in washed river sand into 50-cm-long sections of 15-cm-diameter corrugated plastic pipe. Three pipe sections containing either one or two seedlings of the same species were tied together and placed on 30 x 45 cm fiberglass trays. Ten trays were set on the upper level of each bench to include two donor densities of four different hardwoods plus two sand-only controls. During the summer, greenhouse temperatures were allowed to fluctuate daily between 18 and 35°C with an average daily maximum of 32°C. Minimum temperatures were reduced to 12°C from November 1991 to April 1992 to harden-off the hardwood seedlings before repeating the experiment. Dead seedlings in the donor units were replaced in April 1992. Because of the high mortality and lack of any treatment responses, pawpaw seedlings were replaced by black walnut seedlings for the 1992 experiment.

Holes were drilled near the front edge of 40 fiberglass tray before cementing in PVC couplers with elbows. A 20- to 30-cm-long segment of 1.3-cm-diameter PVC pipe was added to each elbow in order to drain leachates into the center of a receiver pot or the collection jug. Four-liter, semi-transparent plastic jugs were used to collect excess leachates. These jugs were filled with unbuffered tapwater before recycling leachate on a semi-weekly schedule from April 18 (Julian date 108) to October 7 (Julian date 280) in 1991 and from April 15 (Julian date 106) to September 30 (Julian date 101) in 1992. When recycling, approximately one liter of leachate was poured directly over the dittany foliage and one liter of leachate was added to each donor tube. Once a month, the foliage of the

donor seedlings was rinsed with tapwater and throughfall added to the donor pots. Because leachates were continually recycled and not periodically discarded during each experiment, we monitored the leachates for toxicity by testing acidification (pH) and for salt accumulation (conductivity).

One month after starting the 1991 experiment and during the entire 1992 experiment, 200 ml of 0.25x-strength Hoagland's solution (Machlis and Torrey 1956) were added weekly to each pot of dittany to provide an excess of essential nutrients within each experimental unit. In 1991, this corrected the visible symptoms of a nitrogen deficiency. No other visible symptoms of nutrient deficiencies were observed in either the dittany or donor seedlings. To reduce algal growth, the pots and trays of each experimental unit and the collection jugs were covered with aluminum foil.

Leaves of each donor species were collected from a local stand and added monthly to the top of each donor pot during the first 3 months of each experiment. Approximately 300 $cm^2$ of mature leaves (measured with a leaf area meter) of the same species as the donor seedlings were added to the low density donor pots each month. Double this amout was added to the high density donor treatment pots each month. This amount of leaves for each addition is equivalent to the leaf surface over the surface of each pot of dittany from hardwood canopies with an LAI of 1.5 and 3.0, respectively.

Approximately 2, 4, and 6 months after the start of each experiment, the number of leaves on each donor seedling was estimated. Several representative leaves of each species were removed to find the average leaf area. Stem volumes were estimated as the sum of basal diameter squared times height of each stem. Approximately bi-monthly after the start of each experiment, leachate samples from each experimental unit were collected to measure pH and conductivity.

Plant height of each dittany stem was measured bi-weekly until elongation ceased. In addition, a pair of leaflets near the apex of each stem was marked and shoot elongation of the two axillary shoots from this node was measured bi-weekly until flower initiation in early July. Each stem was monitored semi-weekly to determine the Julian date for the first open flowers. An attempt was made to count total number of flowers per stem each week; however, number of flowers were estimated when stems had over 500 open flowers. At the end of each growing season, stem height and number of axillary shoots were measured before plants were cut off to determine above ground stem dry weight (60°C).

The experiment was set up as a 5 x 2 randomized complete block design with 4 blocks of 10 experimental units. In 1991, each block consisted of a factorial arrangement of five donor species (sand-only control, sugar maple, white oak, flowering dogwood, or pawpaw) at two densities. The two sand-only controls were randomly assigned to low and high density within each block. In 1992, the pawpaw treatment was replaced with a black walnut treatment. Means were determined for each experimental unit for stem height, axillary shoot number and length per stem, stem dry weight, date of first flower, number of open flowers, and estimated total flower production. Means were subjected to analysis of variance as a randomized complete block design with four blocks of a 5 x 2 factorial arrangement of treatments for donor species and donor density (SAS Institute Inc. 1988). To determine were significant differences existed among donor species, donor density, or their interaction, the 5% t-test value was calculated from the Type III mean sums of squares (population variance) for doing multiple comparisons.

## RESULTS AND DISCUSSION

Donor seedling survival ranged from a low of 28% for the pawpaw seedlings to 100% for the sugar maple seedlings (Table 1). Because of the low seedling survival, pawpaw effects are probably in response to decomposition of the leaves added to the donor pots and not to any chemicals produced by the 1+0 seedlings. Responses to sugar maple, white oak, flowering dogwood, and black walnut could result from chemicals released from the decomposing leaves and/or from chemicals produced by the planted seedlings. With the frequent recycling of leachates and addition of Hoagland's solution, most of the planted hardwood seedlings showed rapid shoot and leaf growth, except for pawpaw. Many white oak seedlings showed multiple flushes with gradually expanding leaf areas until the experiment was ended each year.

Table 1. Survival, estimated foliage area, and final stem volume of donor seedlings during the two experiments

| Hardwood donor spp. | Number planted | Survival | Estimated foliage area/seedling Week 8 | Week 16 | Week 22 | Stem volume |
|---|---|---|---|---|---|---|
| | | - % - | - - - - - - $cm^2$ - - - - - | | | - $cm^3$ - |
| 1991 EXPERIMENT: | | | | | | |
| White oak | 36 | 72 | 513 | 1,144 | 1409 | 13 |
| Sugar maple | 36 | 100 | 1,363 | 2,659 | 2,073 | 64 |
| Dogwood | 36 | 81 | 476 | 1,959 | 1,690 | 85 |
| Pawpaw | 36 | 28 | 40 | 119 | 57 | 3 |
| 1992 EXPERIMENT: | | | | | | |
| White oak | 36 | 89 | 351 | 568 | —— | — |
| Sugar maple | 36 | 100 | 1,008 | 2,366 | —— | — |
| Dogwood | 36 | 75 | 555 | 1,690 | —— | — |
| Black walnut | 36 | 100 | 20 | 390 | —— | — |

Leachate conductivity was higher in experimental units containing hardwood seedlings than in the sand-only controls suggesting that organic compounds were being released from the seedlings and decomposing leaves (Table 2). During the first and second year, leachate conductivity was highest for sugar maple which also had the greatest shoot growth of the tested hardwood species. During the 1991 experiment, leachate conductivity was similar for the white oak and flowering dogwood which were higher than that for the pawpaw and the sand-only control (Kobe 1992). During the 1992 experiment, recycled leachate conductivities were consistently lower than during the 1991 experiment. One possible explanation is that recycled leachates, which were discarded at the end of the 1991 experiment, resulted in a loss of exchangeable nutrients from the donor and receiver pots that were not replaced in the 1992 experiment.

Leachate conductivity in the 1991 experiment was negatively correlated with the estimated weight of the soil core surrounding the dittany plants ($r = -0.31$, 39 df). This reinforces the conclusion that increased conductivity resulted from chemicals released from the decomposing leaves and the hardwood seedlings. The negative correlation also suggests that released compounds were being bound to the soil particles or that microorganisms within the soil core were decomposing the compounds responsible for increasing leachate conductivity.

Minor differences in leachate pH were found among the donor treatments during each growing season (Table 2). Lack of differences suggests that nutrient toxicities did not develop from the addition of nutrient solutions or continued recycling of the leachates. We have no explanation why the recycled leachate pHs were more alkaline in the 1992 experiment than the 1991 experiment. We assume that nutrients lost after discarding the recycled leachates from the 1991 experiment or washed from the donor pots when they were watered during the winter may have contributed to the increased pH of leachates in the 1992 experiment.

No differences in dittany stem height were found in the 1991 experiment in response to the donor species, density of the donor seedlings, or their interaction (Table 3). Most of the height growth occurred during the first 8 weeks and may have occurred before allelochemicals started to accumulate in the recycled leachates (Figure 2). Mean height of dittany the first year ranged from 38 cm when treated with white oak leachates to 49 cm when exposed to leachates from sand-only (Kobe and others 1992). Similar responses were observed the second year when mean height ranged from 47 cm when exposed to white oak or black walnut leachates to 52 cm when exposed to sand-only leachates. In both years, mean dittany stem height when exposed to sugar maple leachates was statistically the same as when exposed to any other leachate.

Table 2. Mean pH and conductivity of recycled leachates during the 1991 and 1992 experiments

| Donor species | Donor density[1] | pH | | Conductivity | |
|---|---|---|---|---|---|
| | | 1991 | 1992 | 1991 | 1992 |
| | | | | —uSiemens— | |
| Sand-only | High | 7.76[2] | 8.38 | 728 | 692 |
| | Low | 7.76 | 8.45 | 740 | 680 |
| White oak | High | 7.83 | 8.37 | 848 | 787 |
| | Low | 7.77 | 8.61 | 834 | 717 |
| Sugar maple | High | 7.77 | 8.42 | 1115 | 955 |
| | Low | 7.83 | 8.55 | 933 | 876 |
| Dogwood | High | 7.85 | 8.54 | 815 | 896 |
| | Low | 7.78 | 8.58 | 896 | 815 |
| Pawpaw | High | 7.72 | — | 765 | — |
| | Low | 7.62 | — | 712 | — |
| Black walnut | High | — | 8.40 | — | 764 |
| | Low | — | 8.37 | — | 680 |
| 5% t-test values | | 0.35 | 0.94 | 171 | 141 |

[1]The low densities consisted of three donor seedlings and three additions of 300 cm$^2$ of mature leaves of the same species to the litter layer. The high donor densities had double this amount for each donor species.
[2]Mean value of 12 samples in 1991 and 1992.

Table 3. Mean stem length, number of axillary shoots, stem dry weight, Julian date for flower initiation, and number of open flowers during week 17 for dittany plants exposed to leachates produced by six hardwoods at two plant densities

| Donor species | Donor density[1] | Height | | Number of lateral shoots | | Stem dry weight | | Flower init. | Week 17 flowers |
|---|---|---|---|---|---|---|---|---|---|
| | | 1991 | 1992 | 1991 | 1992 | 1991 | 1992 | 1991 | 1991 |
| | | — cm — | | — # — | | — g — | | day | # |
| Sand-only | High | 46.0[2] | 50.9 | 4.0 | 12.3 | 7.1 | 13.1 | 183 | 421 |
| | Low | 48.9 | 52.8 | 6.0 | 12.6 | 8.5 | 13.2 | 198 | 546 |
| White oak | High | 41.7 | 48.5 | 4.4 | 12.6 | 5.9 | 11.9 | 198 | 418 |
| | Low | 47.5 | 41.0 | 4.6 | 10.1 | 4.3 | 10.2 | 186 | 206 |
| Sugar maple | High | 42.0 | 52.7 | 4.6 | 10.4 | 5.3 | 11.2 | 198 | 425 |
| | Low | 44.4 | 49.2 | 5.8 | 12.6 | 6.4 | 10.8 | 200 | 413 |
| Dogwood | High | 42.3 | 44.3 | 5.4 | 13.4 | 7.2 | 13.7 | 191 | 651 |
| | Low | 45.0 | 52.7 | 5.0 | 11.5 | 6.7 | 9.9 | 193 | 594 |
| Pawpaw | High | 45.4 | — | 5.0 | — | 8.4 | — | 193 | 757 |
| | Low | 45.3 | — | 5.4 | — | 8.0 | — | 194 | 596 |
| Black walnut | High | — | 46.2 | — | 4.6 | — | 13.5 | — | — |
| | Low | — | 48.1 | — | 13.7 | — | 11.2 | — | — |
| 5% t-test values | | 8.6 | 7.6 | 2.1 | 3.0 | 2.8 | 4.9 | 15 | 376 |

[1]The low densities consisted of three donor seedlings and three additions of 300 cm$^2$ of mature leaves of the same species to the litter layer. High donor densities had double this amount for each donor species.
[2]Mean value of 4 replications with 2 stems each in 1991 and 3 stems each in 1992.

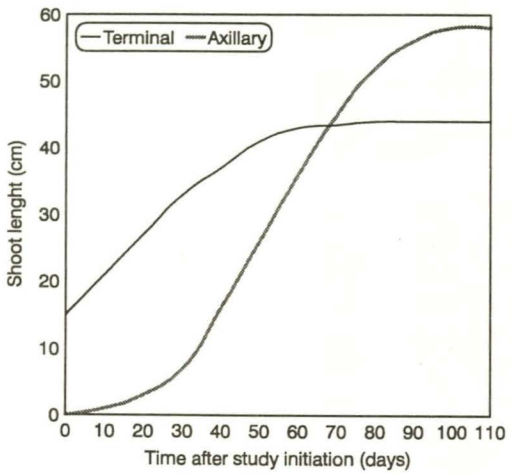


Figure 2. Average terminal and axillary shoot elongation during the first 16 weeks after initiating 1991 study.

Most of the axillary shoot growth occurred 4 to 12 weeks after initiating the study (Figure 2). We found no differences for axillary shoot growth among the donor species, their density, or possible interaction (Kobe 1992). Because axillary growth occurs later than the growth of the main stem, we had expected axillary shoot growth to be a more sensitive indicator of the increasing amounts of recycled compounds in the leachates. One problem with using axillary shoot growth from only one node near the apex to detect allelochemicals was the wide plant-to-plant variation. Future studies should measure total axillary shoot lenght at the end of the experiment or increase the number of replications to reduce the population variance.

Similar responses for height growth and axillary shoot growth were found in the 1992 experiment when pawpaw seedlings were replaced with black walnut seedlings (Table 3). Interesting, when dittany plants were exposed to leachates containing juglone, a known allelochemical of black walnut, they produced more axillary shoots per stem than dittany plants exposed to leachates from sugar maple and white oak. Again, no differences were found in dittany growth in response to leachates produced by any of the hardwood seedlings. These results further reinforce the conclusion that allelochemicals did not accumulate rapidly in the recycled leachates or that shoot growth is not a good variable for measuring dittany reponse to allelochemicals.

Exposure to leachates from both sugar maple and white oak seedlings reduced stem dry weight by the end of the first season compared with leachates from the sand-only control and the pawpaw treatment (Table 3). Responses to flowering dogwood leachates were intermediate and not different from any of the other leachates. We found that the stem dry weight of the dittany plants was positively correlated with the estimated soil core volume ($r = 0.44$, 39 df). Dittany plants with the smaller soil cores began to show nutrient deficiencies one month into the 1991 experiment. The addition of Hoagland's solution visually corrected the problem; however, the positive correlation suggests nutrient deficiencies were not fully corrected with plants still obtaining one or more essential elements from the soil core. A similar response was found after the second growing season where exposure to hardwood seedling leachates also did not result in any statistically significant differences in dittany stem dry weight.

Because shoot growth occurred relatively early in both experiments when the amount of dissolved organic compounds was still low, we chose to also examine effects of the leachates on dittany reproduction. In another study, we found that flowering on dittany normally started in the apical region of the upper shoots and rapidly moved basipetally to the lower nodes and that individiual dittany flowers remained open for approximately 12 days before wilting and abscission of the petals. We concluded that we could count flowers bi-weekly and get a reasonable estimate of total flower production.

In the 1991 experiment, minor differences were found as to the Julian date when the first flowers appeared on each plant (Table 3). Flower initiation began shortly after terminal shoot elongation stopped and before axillary shoot

elongation ceased or approximately 12 weeks after the study was initiated. Most plants had more than 500 open flowers 4 to 8 weeks after flower initiation. Analysis of the number of open flowers 17 weeks after initiating the study showed dittany plants exposed to white oak leachates had fewer flowers than dittany plants exposed to dogwood or pawpaw leachates (Table 3). The number of flowers on plants exposed to sugar maple leachates did not differ from that of any of the other hardwood leachates or sand-only control.

We estimate that many plants produced nearly 2,000 flowers over a 10-week period (Figure 3). The large number of flowers per plant in our study compared to the number of flowers found on plants under forest conditions, suggests our plants in the greenhouse were under little or no stress. No seeds were produced, so we could not determine if hardwood leachates affected seed production. It is unknown whether the high number of flowers occurred because of the excellent growing conditions or because early flowers did not set seed.

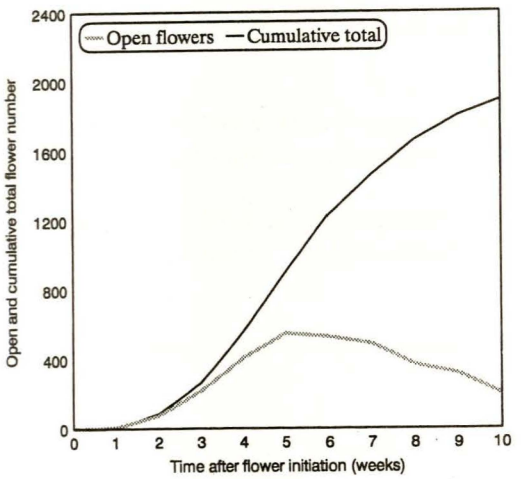


Figure 3. Number of open flowers and cumulative totals during the 10 weeks following initiation.

Exposure to leachates from several hardwoods reduced stem dry weight and altered flower production; however, doubling the number of donor seedlings did not alter these responses. There was an indication that white oak leachates were more toxic than sugar maple leachates. If, however, one or more of the hardwood species were producing allelochemicals in significant quantities, we should have observed a donor species x donor density interaction. This interaction was not detected in the 1991 or the 1992 experiment. The results of this study strongly suggest that the loss of dittany in oak-hickory forests following the invasion by sugar maple is not due to allelopathy but probably results from competition for light or the heavy accumulation of leaf litter in the stands.

In conclusion, we could not totally rule out the occurrance of allelopathic interference of hardwoods on understory forbs such as dittany. In other studies moisture and nutrient stresses have been shown to significantly affect plant responses to allelochemicals (Stowe 1979, Rink and Van Sambeek 1985, Van Sambeek and others 1996). Additional studies designed to vary light intensity or moisture availability are needed to better separate interference into its components of competition and allelopathy.

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# RESPONSE OF POTTED NORTHERN RED OAK AND HAY-SCENTED FERN TO ADDITIONS OF CALCIUM, MAGNESIUM, POTASSIUM, AND PHOSPHORUS

David H. Hart and William E. Sharpe[1]

Abstract: The objective of this study was to determine if addition of Ca, Mg, K, and P to an extremely acidic forest soil would increase early northern red oak (*Quercus rubra* L.) seedling growth. The O through B2 horizon of a Dekalb soil from Pennsylvania's Bald Eagle State Forest was placed in plastic cores and utilized as a growth medium for northern red oak seedlings in a greenhouse environment. After 150 days, fertilization with the selected nutrients Ca, Mg, K, and P increased greenhouse seedling height, leaf area, leaf dry weight, stem diameter, stem dry weight, and tap root diameter. Fertilization also resulted in increased foliar concentrations of P, K, Ca, and Mg and decreased foliar concentrations of Mn and Al. Hay-scented fern (*Dennstaedtia punctilobula* (Michx.) Moore), grown in the same containers with the northern red oak seedlings, demonstrated the potential to be an effective sink for applied fertilizers containing Ca, Mg, K, and P. These results imply that on extremely acidic sites where soils are nutrient deficient, fertilization with Ca, Mg, K, and P may enhance early northern red oak seedling growth.

## INTRODUCTION

There has been considerable debate over the decline of temperate-zone forests in Europe and North America. In Pennsylvania, it has become quite clear that adequate advance regeneration of hardwood forests is lacking in many areas of the state (McWilliams and others 1993). One commercially valuable hardwood timber species experiencing this regeneration problem is northern red oak. Chapman and Gower (1991) reported that northern red oak is rapidly disappearing from natural forests in the eastern United States due to the high demand for red oak lumber and the inability of forest managers to consistently promote its natural regeneration.

Possible causes of failed northern red oak regeneration include soil acidification and reduced nutrient availability (Lyon 1995), interference by hay-scented fern (Kolb and others 1989), and herbivory by deer (Stout and others 1995). Because soil acidification and dense hay-scented fern cover could both have detrimental effects on red oak seedling growth, a hypothesis combining the two would seem presumable in some areas where a regeneration problem exists. Soil acidification increases leaching of exchangeable plant nutrients, reduces the availability of phosphorus, and increases the solubility of other elements, some of which are phytotoxic (Cowling and Linthurst 1981). The possibility of red oak growth declines as a result of Al toxicity on Northeastern Spodosols was noted by Joslin and Wolfe (1989), and Sharpe and others (1993) reported that the loss of Ca from forest soils as a consequence of acidic deposition may be important in the height growth of red oak seedlings. On the other hand, the distribution of calcifugous fern types is associated with acid soils where a large amount of aluminum is available, and hay-scented fern, in particular, also may be adapted to the relatively high availability of manganese in acid soils (Hou 1950).

Lorimer (1989) noted that techniques for enhancing the early growth of northern red oak seedlings will hopefully reduce the need for tall oak advance regeneration. One such technique is the placement of fertilizer in the planting hole as part of a regeneration method where the young plant is ensured a sufficient supply of nutrients during the first years of growth (Baule and Fricker 1970). Tomlinson and Tomlinson (1990) reported that fertilization and liming can effectively revitalize forest ecosystems and work to alleviate forest damage associated with nutritional disturbance of soils by atmospheric deposition.

[1]Former Graduate Assistant and Professor of Forest Hydrology, respectively, School of Forest Resources and Environmental Resources Research Institute, The Pennsylvania State University, University Park, PA 16802.

The objective of this research was to determine if addition of Ca, Mg, K, and P to an extremely acid soil from a site with failed northern red oak regeneration would improve northern red oak seedling growth, with and without competition from hay-scented fern.

## METHODS

The soil used in this greenhouse study was a Dekalb (loamy-skeletal, mixed mesic Typic Dystrochrept) from Pennsylvania's Bald Eagle State Forest (USDA Soil Conservation Service 1985). Soil from this site was selected because it was a piece of uncut forest within deer exclusion fencing, and although both northern red oak and chestnut oak trees were located in the overstory, there were no northern red oak seedlings in evidence. A plastic protection collar with a 15 cm diameter was used as a reference device to remove adjacent and intact 15 cm diameter circles of the organic soil layer. The mineral soil underneath the removed organic layers was collected down to a depth of 25 cm (B2 horizon) and sifted through a 1 cm sieve.

Forty-eight clear plastic cores, covered with aluminum foil to prevent light entry, were arranged on a greenhouse table. The cores were 15 cm in diameter, 25 cm in height, and had a 0.6 cm hole in the bottom for drainage. To aid in drainage, a layer of small pebbles was spread evenly across the bottom of the cores, and cheesecloth was placed on top to prevent soil from eroding through. Twenty-four of the cores were fertilized in the following manner: mineral soil, placed in a sample core to a height of 21 cm, was thoroughly mixed with 10 g of crushed dolomitic limestone (Baule and Fricker 1970); a portion of this soil was used to make a 2 cm high layer above the cheesecloth; next, 20 g of granular phosphorus in the form of $P_2O_5$ (0-44-0) was placed to one side of the core (Baule and Fricker 1970); subsequently, the rest of the mineral soil was added, and an intact 15 cm diameter organic layer was placed on top; lastly, around the outside of the organic layer, 5 g (224.5 kg/ha) of 60% KCl fertilizer (0-0-60) was distributed (Walker 1956). For the 24 unfertilized cores, everything was done in the same manner except that no fertilizer was added.

A hole was cut through the middle of the organic layer, and germinated northern red oak seeds from the same mother tree located on the Penn State University Campus were placed on top of the mineral soil-one seed per core. Additionally, 8-10 cm long hay-scented fern rhizomes were added to 12 of the fertilized and to 12 of the unfertilized cores. These rhizomes did not grow and were replaced 40 days into the study with two rhizomes of similar length placed underneath the organic layer on each side of the acorn.

The greenhouse lights (400 watt High Pressure Sodium) were set at 16 hours per day (DeWald and others 1990), and a hygrothermograph was setup along with a Taylor Model 5458 maximum/minimum thermometer to monitor the temperature on the table holding the cores. During the time period of the study, the temperature on the greenhouse table ranged from 9-33°C (48-91°F). A 20% black shadecloth was placed over the seedlings for the entire duration of the study. The seedlings received 125 ml of deionized water twice weekly for the first 90 days of the study; for the remaining 60 days, the amount was increased to 150 ml twice weekly (Lyon 1994).

Height growth of the seedlings was measured at 30 day intervals over the course of the study. After 150 days of growth, the seedlings and ferns were sacrificed, and additional measurements were collected. Fern fronds were cut at ground level, oven dried at 85°C for 33 hours, weighed to the nearest 0.01 g on a Mettler PR1200 balance, and frozen for 10 weeks prior to chemical analysis. Oak seedling leaf areas were measured to the nearest 0.01 $cm^2$ using a LI-3100 Leaf Area Meter (Sharpe and others 1993) before the leaves were washed with deionized water. The leaves were then placed in an oven at 85°C (Joslin and Wolfe 1989) for 36 hours. Upon removal, they were weighed to the nearest 0.01g and frozen for 10 weeks prior to chemical analysis (Sharpe and others 1993). Starrett digital calipers were used to measure oak seedling diameters to the nearest 0.01 mm at a height just above the root collar (DeWald and others 1990). Seedling stems were cut from tap roots at the root collar, and the root diameters were taken just below the root collar (Sharpe and others 1993). The seedling and fern root systems were carefully removed from the cores and washed over USA Standard Testing 2 mm and 0.5 mm sieves to retain any fine roots that broke off (Safford 1974). Oak seedling lateral roots were removed from tap roots with scissors. Total tap root lengths were measured by using a string to trace the path of each root and measuring the string to the nearest 0.1 cm. The seedling lateral and tap roots were placed in an oven at 85°C for 36 hours and weighed to the nearest 0.01 g

upon removal (DeWald and others 1990). Fern rhizomes and roots and red oak stems were oven dried at 85°C for 48 hours and weighed to the nearest 0.01 g upon removal.

For chemical analysis of seedling leaves and fern fronds, samples were submitted to the Agricultural Analytical Services Laboratory Plant Analysis Program (Penn State University), and concentrations were obtained by an inductively coupled plasma emission spectrometer according to methods outlined by Dahlquist and Knoll (1978). Calibration of the spectrometer was performed using reference samples which were obtained from the National Institute of Science and Technology (NIST) and with in house check samples. Analysis included concentrations of P, K, Ca, Mg, Mn, Fe, Cu, B, Al, Zn, and Na; however, only the results concerning P, K, Ca, Mg, Mn, and Al are discussed.

At the conclusion of the study, soil samples were collected from the intact organic soil surrounding the seedlings and from the previously mixed mineral soil at a depth of 10-11 cm beneath the organic soil layer. Thus, one organic soil sample and one mineral soil sample was collected from each core. Soil analysis was completed on both organic and mineral soils for exchangeable Mn and Al using a 0.01M $SrCl_2$ extraction (Joslin and Wolfe 1989). Concentrations were obtained using an Instrumental Laboratory Video 22 aa/ee spectrophotometer. Additionally, pH, available P, and exchangeable concentrations of K, Mg, and Ca were obtained from the Agricultural Analytical Services Laboratory (Penn State University) according to methods outlined in North Dakota State University (1988).

Since data were not normally distributed, nonparametric procedures were used in the analyses. Kruskal-Wallis one-way analysis of variance was used to test for differences among 4 treatments: fertilized with fern, unfertilized with fern, fertilized without fern, and unfertilized without fern (Daniel 1978). When treatment effects were significant (e.g, $P < 0.05$), multiple-comparison procedures (Dunn 1964) were used to detect differences between individual treatments. Kruskal-Wallis statistics (H) were corrected for ties, and an experimentwise error rate of 0.20 was used for multiple-comparison procedures (Kurtz and others 1965).

## RESULTS

### Soil Chemistry

In the organic soil chemistry comparison (Table 1), changeable Mn concentrations for the fert. & no fern treatment were significantly higher than those for the fert. & fern treatment. Exchangeable Al concentrations were not significantly different among treatments. The fert. & no fern and the no fert. & fern pH values were significantly higher than the fert. & fern pH values. The fert. & no fern available P concentration was significantly higher than the no fert. & fern concentration. Exchangeable K and Mg concentrations were significantly higher in the fertilized than the unfertilized treatments while the exchangeable Ca level was significantly higher for the fert. & no fern treatment than the no fert. & no fern treatment.

In the mineral soil (Table 1), pH was significantly higher in the fertilizer treatments than the no fert. & fern treatment. The fert. & fern pH was also significantly greater than the no fert. & no fern pH. Exchangeable Mn and Al levels were significantly higher for the unfertilized than the fertilized treatments. Exchangeable K, Mg, and Ca levels were significantly higher for the fertilized treatments than the unfertilized treatments. The available P was not significantly different among treatments due to the methods of fertilization and sampling. Overall, the mineral soil chemistry results indicated that the fertilizer treatments impacted the mineral soil through a higher pH, increased levels of exchangeable K, Mg, and Ca, and decreased concentrations of exchangeable Mn and Al.

Table 1. Median organic and mineral soil chemical parameters at the conclusion of the 150 day greenhouse study

| Variable | Fert. & Fern | No Fert. & Fern | Fert. & No Fern | No Fert. & No Fern | H Test Statistic |
|---|---|---|---|---|---|
| Organic Soil | | | | | |
| Mn (mg $kg^{-1}$) | 63.0a | 72.1ab | 101.6b | 68.8ab | 7.06 |
| Al (mg $kg^{-1}$) | 1.5a | 4.4a | 3.3a | 5.2a | 1.73 |
| pH | 4.0a | 4.3b | 4.3b | 4.2ab | 9.93 |
| P (mg $kg^{-1}$) | 26ab | 21a | 34b | 24ab | 8.45 |
| K (meq/100 g) | 3.68a | 0.55b | 4.24a | 0.39b | 28.86 |
| Mg (meq/100 g) | 0.7a | 0.6b | 1.3a | 0.5b | 30.53 |
| Ca (meq/100 g) | 1.5ab | 1.5ab | 2.0a | 1.5b | 9.13 |
| Mineral Soil | | | | | |
| Mn (mg $kg^{-1}$) | 1.0a | 4.4b | 0.5a | 3.7b | 35.34 |
| Al (mg $kg^{-1}$) | 0.4a | 12.7b | 0.4a | 12.2b | 36.86 |
| pH | 5.9a | 4.6b | 5.8ac | 4.6bc | 18.46 |
| P (mg $kg^{-1}$) | 36a | 26a | 36a | 31a | 3.74 |
| K (meq/100 g) | 0.57a | 0.11b | 0.56a | 0.09b | 31.05 |
| Mg (meq/100 g) | 1.5a | 0.2b | 1.4a | 0.2b | 35.19 |
| Ca (meq/100 g) | 2.0a | 0.5b | 2.0a | 0.5b | 36.41 |

Different letters within a row indicate significant differences among treatments at an $\alpha \leq 0.05$ according to Dunn's (1964) multiple-comparison procedures.

Seedling Foliar Chemistry

Foliar chemistry comparisons (Table 2) revealed that seedling foliar P and Mg concentrations were significantly greater in the fertilized than the unfertilized treatments. Foliar K concentration was significantly greater in the fert. & fern treatment than the no fert. & fern and no fert. & no fern treatments. Foliar Ca concentration was significantly higher in the fertilized treatments than the no fert. & no fern treatment, and concentrations of Mn and Al were significantly greater in the unfertilized than the fertilized treatments. In effect, the fertilizer treatments increased the seedling foliar concentrations of the selected nutrients P, K, Ca, and Mg and decreased the foliar concentrations of Mn and Al.

Table 2. Median northern red oak seedling foliar element concentrations at the conclusion of the 150 day greenhouse study

| Element | Fert. & Fern | No Fert. & Fern | Fert. & No Fern | No Fert. & No Fern | H Test Statistic |
|---|---|---|---|---|---|
| P (%) | 0.19a | 0.12b | 0.18a | 0.12b | 24.37 |
| K (%) | 0.95a | 0.73b | 0.84ab | 0.72b | 10.14 |
| Ca (%) | 0.77a | 0.63ab | 0.78a | 0.59b | 11.57 |
| Mg (%) | 0.30a | 0.10b | 0.29a | 0.07b | 34.47 |
| Mn (mg $kg^{-1}$) | 3775a | 11550b | 2950a | 15655b | 25.65 |
| Al (mg $kg^{-1}$) | 29a | 68b | 29a | 80b | 29.47 |

Different letters within a row indicate significant differences among treatments at an $\alpha \leq 0.05$ according to Dunn's (1964) multiple-comparison procedures.

Hay-scented Fern Frond Chemistry

Hay-scented fern frond concentrations of P, K, Ca, and Mg were significantly higher in the fertilized than the unfertilized treatment (Table 3). The concentration of Al was significantly higher in the unfertilized treatment, and the concentration of Mn was not significantly different between the treatments (Table 3). Thus, with the exception of Mn, the hay-scented fern frond chemistry response to the fertilization corresponded with that of the northern red oak seedlings.

Table 3. Median hay-scented fern frond element concentrations at the conclusion of the 150 day greenhouse study

| Element | Fert. & Fern | No Fert. & Fern | H Test Statistic |
|---|---|---|---|
| P (%) | 0.13a | 0.07b | 7.14 |
| K (%) | 3.20a | 1.13b | 10.50 |
| Ca (%) | 0.41a | 0.24b | 9.42 |
| Mg (%) | 0.29a | 0.12b | 10.61 |
| Mn (mg $kg^{-1}$) | 1231a | 1166a | 0.21 |
| Al (mg $kg^{-1}$) | 65a | 141b | 3.88 |

Different letters within a row indicate significant differences between treatments at an $\alpha \leq 0.05$ according to Dunn's (1964) multiple-comparison procedures.

Growth Parameters

Growth comparisons among treatments (Table 4) tended to reflect the changes evidenced in the soil chemistry and foliar chemistry comparisons. The comparison of seedling heights at 150 days showed that the fert. & fern treatment was significantly taller than the no fert. & fern and the no fert. & no fern treatments, and the fert. & no fern treatment was significantly taller than the no fert. & no fern treatment. The fert. & fern treatment leaf area and dry

Table 4. Median northern red oak seedling and hay-scented fern growth parameters at the conclusion of the 150 day greenhouse study

| Growth Parameter | Fert. & Fern | No Fert. & Fern | Fert. & No Fern | No Fert. & No Fern | H Test Statistic |
|---|---|---|---|---|---|
| Seedling | | | | | |
| height (150 d) (cm) | 28.9a | 18.9bc | 27.6ab | 15.0c | 18.07 |
| leaf area ($cm^2$) | 797.9a | 365.4bc | 632.3ab | 343.9c | 17.06 |
| leaf dry weight (g) | 3.95a | 1.92bc | 3.66ab | 1.84c | 15.23 |
| stem diameter (mm) | 6.87a | 6.07ab | 6.97a | 5.14b | 13.44 |
| stem dry weight (g) | 2.24a | 1.63ab | 2.56a | 1.13b | 12.55 |
| tap root diameter (mm) | 10.72a | 8.14ab | 9.73a | 7.38b | 14.30 |
| tap root length (cm) | 47.7a | 43.7a | 47.1a | 58.3a | 1.44 |
| tap root dry weight (g) | 4.51a | 4.53a | 3.94a | 3.81a | 0.44 |
| lateral root dry weight (g) | 1.72a | 1.11a | 1.71a | 1.02a | 0.32 |
| Fern | | | | | |
| frond dry weight (g) | 0.29a | 0.33a | | | 1.02 |
| rhizome & root dry weight (g) | 0.57a | 1.31a | | | 3.23 |

Different letters within a row indicate significant differences among treatments at an $\alpha \leq 0.05$ according to Dunn's (1964) multiple-comparison procedures.

weight were significantly greater than the no fert. & fern and the no fert. & no fern treatments. The fert. & no fern treatment leaf area and dry weight were significantly greater than the no fert. & no fern treatment. Stem diameter and stem dry weight were significantly larger in the fertilized treatments than the no fert. & no fern treatment. A belowground response was only apparent in the tap root diameter which was increased by fertilization. Seedling tap root lengths, tap root and lateral root dry weights, and the lengths and diameters of the 5 longest lateral roots did not significantly differ among any of the treatments. Hay-scented fern frond dry weights and hay-scented fern rhizome and root dry weights also did not significantly differ between the treatments.

## DISCUSSION

### Soil Chemistry

The only fertilizer applied directly upon the organic soil was the 60% KCl fertilizer. Soil analysis results showed that it was still having an impact through increased exchangeable K concentrations in the organic soil at 150 days.

For the mineral soil, the results showed that the 60% KCl fertilizer applied on the surface of the organic soil was impacting the mineral soil through increased exchangeable K concentration at 150 days. The crushed dolomitic limestone application led to significantly higher exchangeable concentrations of Ca and Mg in addition to a pH increase in the mineral soil of the fertilized treatments. Due to the placed application of P fertilizer, the method of sampling could not accurately indicate the amount of available P. The foliar analysis, however, showed that the P fertilization did increase the P available to the northern red oak seedlings. The fertilization significantly decreased the mineral soil exchangeable concentrations of Mn and Al. Since soil Al has been closely and negatively correlated with root system branching and development (Joslin and Wolfe 1989), a decrease in the exchangeable Al concentration might benefit the growth of red oak seedlings.

### Seedling Foliar Chemistry

As indicated by the general trends in the soils, the fertilizer applications increased the foliar concentrations of the selected nutrients P, K, Ca, and Mg and decreased the foliar concentrations of Mn and Al. Over the course of the 150 day study, older leaves on unfertilized seedlings displayed interveinal chlorosis and interveinal stippling. Since precise critical values could not be found for northern red oak, the reported range or minimum foliar element concentrations for other dicotyledonous tree species displaying visual symptoms of nutritional deficiency were used as references to explain the yellowing and the stippling of the unfertilized seedling leaves. A combination of nutrient deficiencies and toxicities appeared to cause the visual symptoms.

The foliar K concentrations in both the fertilized and unfertilized seedlings were above the 0.29-0.66% range that Walker (1956) cited for display of visual deficiency symptoms in red maple. The foliar Ca concentrations in the fertilized and unfertilized seedlings were also above the 0.30% critical value cited by Huttl (1986) for the display of visual deficiency symptoms in European beech. The foliar P and Mg concentrations of the unfertilized seedlings were below the reported critical values, 0.15% for both P and Mg in European beech (Huttl 1986), for the display of deficiency symptoms while their concentrations in the fertilized seedlings were above the critical values. Therefore, these nutrients may have been limiting factors in the unfertilized seedling growth.

Although the foliar Mn concentration was high for all of the treatments, it was extremely high for the unfertilized seedlings. Messenger (1984) noted that 100 to 280 ppm foliar Mn is evident in green and apparently healthy red maple leaves. Bergmann (1992) reported that in most cases, the symptoms caused by toxic manganese concentrations usually are seen on the older leaves first, appearing initially as brown spots (deposits of $MnO_2$) or chlorotic and necrotic lesions, which generally spread from the tips and margins to the center of the leaf and are often accompanied by rolling of the leaf edges. These symptoms were evident on the older leaves of many of the unfertilized seedlings in this experiment. Thus, Mn concentrations were possibly at toxic levels in the foliage of the unfertilized seedlings. Bergmann (1992) reported that the most important factors encouraging the release of $Mn^{2+}$ cations into

the soil solution are low pH values (below 5.5/6.0) and a high reduction capacity of the soil, particularly if it is poorly drained or occasionally flooded.

In review, the foliar literature values suggested that P and Mg deficiencies and Mn toxicity may have been inhibiting factors in the unfertilized red oak seedling growth. For the fertilized seedlings, the foliar element concentrations were above critical values cited in the literature. Thus, the comparison with the literature values indicated that the combination of different fertilizers was effective in meeting the nutritional requirements of the red oak seedlings.

Hay-scented Fern Frond Chemistry

The effects of the fertilizer applications on hay-scented fern frond chemistry appeared to parallel the effects on the red oak seedling foliage. Concentrations of P, K, Ca, and Mg in the fronds increased while that of Al decreased, demonstrating that hay-scented fern has the potential to be a sink for these nutrients. This could be an important factor for fertilization attempts in areas where hay-scented fern ground cover exists. Fertilizer applied on the ground surface may be taken up by the dense matting of fern roots located in or just below the organic layer before it comes in contact with seedling roots. This type of fertilization could inhibit seedling growth by aiding in hay-scented fern growth. Hay-scented fern resources not used in reproductive or photosynthetic efforts may be stored in rhizomes and can facilitate rapid growth in the next growing season (Hammen 1993). Methods of fertilization which get nutrients well down into the mineral soil layers could make the fertilizer less available to the fern roots and more readily available to the oak seedling roots.

Unlike the red oak seedlings however, the concentration of Mn in the hay-scented fern fronds was not significantly different between fertilized and unfertilized treatments. This lack of a difference suggested that the hay-scented fern had some mechanism of limiting Mn uptake. The northern red oak seedlings did not display this ability to limit Mn uptake. If such an Mn uptake limiting mechanism does exist, it could be an important factor in the growth competition between red oak seedlings and hay-scented ferns on extremely acidic forest soils with high levels of available Mn.

Growth Parameters

The results of the height comparison supported the results of the foliar analysis. The fertilized treatments displayed increased seedling height growth at 150 days, indicating that the fertilization methods were successful in aiding northern red oak seedling growth.

Seedling leaf area, leaf dry weight, stem diameter, stem dry weight, and tap root diameter were greater for the fertilized treatments without regard to fern. This fertilizer effect was hypothesized, and the increase in the above ground growth variables corresponded with the nutrient increases in the foliage of the fertilized seedlings. Except for tap root diameter, the below ground northern red oak seedling variables did not display a response to the fertilization. The fern was probably not a significant factor because the northern red oak seedlings were well established when the hay-scented fern rhizomes were planted. This result indicated the possibility for northern red oak seedlings to effectively compete with hay-scented fern when the seedlings are established first and properly nourished with Ca, Mg, K, and P.

The hay-scented fern frond and the rhizome and root dry weights tended to be higher for the unfertilized treatments, but the differences were not significant. However, in the case of the rhizome and root dry weight, the unfertilized median was twice as high as the fertilized median. The opposite was true for the median northern red oak seedling lateral root dry weight which was higher for the fertilized treatment. Although these differences were not significant, they presented the possibility that the fertilizer applications may have resulted in more and/or larger northern red oak seedling lateral roots which may have contributed to smaller hay-scented fern rhizome and root growth.

## SUMMARY

Northern red oak seedling responses to fertilization with the selected nutrients Ca, Mg, K, and P were measured in a greenhouse experiment. Fertilization increased northern red oak seedling height, leaf area, leaf dry weight, stem diameter, stem dry weight, and tap root diameter. Fertilization also resulted in increased foliar concentrations of P, K, Ca, and Mg and decreased concentrations of Mn and Al. Under the conditions of this study, hay-scented fern did not appear to affect northern red oak seedling growth and seemed indifferent to changes in soil Mn availability. However, hay-scented fern demonstrated the potential to be an effective sink for applied fertilizers containing Ca, Mg, K, and P.

Techniques for enhancing the early growth of planted seedlings may have important implications for the regeneration of northern red oak seedlings following timber harvest or other forest disturbance. On sites where extremely acidic soils, high in available Al and Mn but deficient in nutrients, are present, fertilization with Ca, Mg, K, and P by the methods used in this study may enhance early northern red oak seedling growth.

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# LARGE-SCALE COMPARISON OF REFORESTATION TECHNIQUES COMMONLY USED IN THE LOWER MISSISSIPPI ALLUVIAL VALLEY: FIRST YEAR RESULTS

Callie J. Schweitzer[1], John A. Stanturf[1], James P. Shepard[2],
Timothy M. Wilkins[3], C. Jeffrey Portwood[4], and Lamar C. Dorris, Jr.[3]

Abstract: In the Lower Mississippi Alluvial Valley (LMAV), restoring bottomland hardwood forests has attracted heightened interest. The impetus involves not only environmental and aesthetic benefits, but also sound economics. Financial incentives to restore forested wetlands in the LMAV can come from federal cost share programs such as the Conservation Reserve Program and the Wetlands Reserve Program (WRP), which provide easements and funding to cover planting costs. Forestry led Mississippi agriculture production in 1995, testimony to the importance of timber. This project is designed to test four restoration techniques and includes methods outlined by both federal program guidelines and by industry. The four techniques being tested are: (1) Eastern cottonwood (*Populus deltoides* Marsh.) as a nurse crop for Nuttall oak (*Quercus nuttallii* Palmer), (2) planting bareroot Nuttall oak seedlings, (3) direct seeding Nuttall oak acorns and, (4) natural, old field succession. A former farmland in Sharkey County, MS, was chosen as the study area because it represents edaphic and hydrologic conditions frequently encountered in economically marginal cropland in the LMAV. A randomized complete block design was used with three replications. Block size was 32 hectares, allowing for four 8 ha treatment plots. All planting was completed by March 1995, except for the interplanting of oak in the cottonwood, which will occur in year 3. Acorns were sown in May 1995. Sample plots were measured in fall 1995. Survival after one growing season averaged 92% for cottonwood, 63% for planted oak seedlings, and 11% for oak germinants. No trees were found in the natural succession plots. Number of stems per ha averaged 793 for cottonwood, 467 for seedlings and 284 for germinants. Average height of cottonwood after the first growing season was 2.1 meters, seedling height was 0.4 m, and germinant height was 0.1 m. There was variability between blocks, and seedling damage by rodents was prominent. Tree planting on land enrolled in the WRP is considered successful for wetland restoration purposes if 309 live trees per ha are present after three growing seasons. After 1 year, both the cottonwood and oak plantings resulted in stocking that could meet these standards. Although the stocking mandated by WRP was not met for the direct seeded oak after one growing season, delayed germination may increase stocking. In addition to providing wildlife habitat, the high survival rate and rapid growth of the cottonwood will enable harvest in 10 years, generating income in a relatively short time period. Further evaluation of tree growth and survival under these restoration techniques will allow for a more definitive conclusion on their feasibility and attractiveness to landowners.

## INTRODUCTION

The Federal Agriculture and Reform Act of 1996, commonly called the Farm Bill, was recently passed by Congress and contains some provisions to promote afforestation in the lower Mississippi alluvial valley. Both the Conservation Reserve and Wetland Reserve Programs were reauthorized; these programs are instrumental in providing landowner incentives to plant trees. In addition, the Farm Bill authorized $50 million to the Wildlife Habitat Incentives Program for cost-share payments to landowners to develop wildlife habitat, and tree planting is included. Nevertheless, the potential exists for thousands of hectares of additional land to be planted to trees beyond that included in federal programs. One way to realize the potential is to develop alternative reforestation techniques that

[1] Reforestation Specialist and Project Leader, respectively, USDA Forest Service, Center for Bottomland Hardwood Research, Southern Research Station, Stoneville, MS.
[2] NCASI, Gainesville, FL.
[3] Refuge Manager and Refuge Forester, respectively, USDI Fish and Wildlife Service, Yazoo National Wildlife Refuge Complex, Hollandale, MS.
[4] Crown Vantage Inc., Fitler Managed Forest, Rolling Fork, MS.

offer more immediate financial returns to landowners than is provided by conventional sawlog rotations or by wildlife habitat/wetland restoration techniques.

The most commonly used reforestation approach has been to plant one to three overstory tree species, usually oaks (*Quercus* spp.). Artificial regeneration is accomplished by either sowing acorns or other hard mast species, or by planting bareroot seedlings. On former agricultural lands, machine planting of seedlings or sowing of acorns or other hard mast is common. This project was designed to test one alternative reforestation technique that combines a faster growing species, cottonwood (*Populus deltoides* Marsh.) and a slower growing species, Nuttall oak (*Quercus nuttallii* Palmer), and contrast this technique with more traditional approaches of planting bareroot seedlings or direct seeding of acorns of Nuttall oak. The control treatment for this study is "do nothing" and allow natural, old field succession to occur.

Compared to other reforestation techniques, one advantage of using the cottonwood nurse crop may be the creation of a more favorable microclimate for oak growth and survival. Obviously, we cannot test this until later in the study. The early growth of cottonwood allows for the rapid establishment of a forest canopy. The advantage of this canopy is that it may lend itself to accelerating natural succession by attracting birds and small mammals that are vectors for dispersal of heavy seeds.

The major disadvantage of pure cottonwood plantations to wildlife may be the paucity of hard mast. Although some may also feel that cultivation works against restoration goals, it has been found that wildlife importance values for all wildlife food plants in several cottonwood plantations studied peaked in the fourth, fifth and sixth growing seasons (Wesley and others 1981). The planting scheme under study here will provide for hard mast, and cultivation cessation after establishment will hopefully serve to promote plant establishment which would benefit wildlife and restoration in the later growing seasons.

In addition to providing wildlife habitat, the high survival rates and rapid growth of cottonwood will enable harvest in 10 years, generating income in a relatively short time period. Stanturf and others (in review) conducted, as an example, a financial analysis for a private landowner of a cottonwood plantation and found that the cottonwood provided a cash flow over the rotation, and a positive internal rate of return. Data collected from this study will allow for future cost analysis with the additional returns of the oak together with the cottonwood.

## METHODS

The study is located on a tract of abandoned farmed wetland in Sharkey County, MS, 2.5 km east of the community of Anguilla, approximately 1 km south of State Highway 14, and immediately north of the Delta National Forest. This land was transferred to the USDI Fish and Wildlife Service from the Farmers Home Administration in 1993, and is administered by personnel at the Yazoo National Wildlife Refuge Complex.

A recent soil survey of the tract conducted by the Natural Resources Conservation Service indicates that the soil is of the Sharkey series, a heavy clay with shrink-swell properties (very fine, montmorillonitic, nonacid, thermic Vertic Haplaquepts; Floyd Wood, NRCS, Jackson, MS, pers. comm. Dec. 1995). The Sharkey series consists of poorly drained, clayey soils formed in fine textured sediment in slack water areas along the Mississippi River. The shrink-swell nature typical of clays with montmorillinitic mineralogies results in 2-10 cm wide cracks up to 1.5 m deep that form under dry conditions, and close upon wetting. The hydrologic and edaphic conditions of our study site typify most of the adandoned agricultural land available for, and undergoing, reforestation in this region.

The experimental design follows a randomized complete block with three replicates located in different portions of the tract. Treatment plots are approximately rectangular and 8.1 hectares in size. Treatment plots were established in October, 1994 (Blocks II and III) and February, 1995 (Block I). Azimuths were measured with a transit and distances with a 30.5 meter tape. Plot corners were permanently marked with steel fence posts. The assignment of treatment to plots was done using a random numbers table, and is summarized in Table 1. The cottonwood/oak plots have a 9 meter buffer around them for machine turn rows, as they will be cultivated for 2 years. All other plots share borders.

Table 1. Treatment plot assignment in each block

| | BLOCK | | |
|---|---|---|---|
| TREATMENT PLOT | I | II | III |
| 1 | PLN | SOW | NUR |
| 2 | NAT | NAT | SOW |
| 3 | NUR | PLN | NAT |
| 4 | SOW | NUR | PLN |

PLN = Plant bare-root Nuttall oak seedlings (1-0 stock)
NAT = Natural old field succession
NUR = Cottonwood/Nuttall oak intercrop
SOW = Direct seeded Nuttall oak acorns

Agricultural production on the study site ended in September, 1994. Site preparation consisted of double disking all treatment plots (including the natural succession treatment) in October, 1994. Acorns were collected on the Delta National Forest, placed in water and non-viable acorns that floated were discarded. Acorns were then stored in ventilated polyethylene bags at 1.7° C. The Fish and Wildlife Service conducted the planting/sowing using their standard equipment and techniques. In May 1995, acorns were machine sown at 1.1 m X 3.7 m spacing, with one acorn placed at each planting spot. Oak seedlings (1-0 stock) were obtained from Fratesi Nursery, Leland, MS, (seed source was the Delta National Forest). Seedlings were machine planted in March 1995 at 3.7 m X 3.7 m spacing.

Cottonwood was chosen as a nurse crop to establish Nuttall oak because of its economic value and its suitability, under cultivation, to Sharkey soils (Stanturf and others, in review, Stanturf and Shepard 1995). Cottonwood cuttings were planted by Crown Vantage Inc., following their operational procedures (Table 2). Four cottonwood clones, ST66, ST72, ST75 and S7C1 were hand planted in pure blocks at 3.7 m X 3.7 m spacing in March 1995. Cultivation will keep the cottonwood practically weed-free for two growing seasons. In March 1997, Nuttall oak seedlings will be interplanted at 3.7 m X 7.3 m spacing (i.e., between every other cottonwood row). Pulpwood harvests will remove most of the cottonwood at year 10 and coppice regrowth at year 20; at year 20 the oaks will be fully released, although some cottonwood stems will be retained beyond year 20 to increase diversity.

Table 2. Schedule of operations for cottonwood/nuttall treatment

| | |
|---|---|
| October 1994 | Two-pass site preparation disking<br>Row establishment and liquid nitrogen applied in trenches at 112 kg N $ha^{-1}$ |
| March 1995 | Plant cottonwood |
| March 1995 | Spray herbicide in bands over dormant cuttings<br>(oxyfuorfen @ 0.26 kg $ha^{-1}$ + glyphosate @ 1.4 kg $ha^{-1}$) |
| May 1995 | One pass disking, followed 2 weeks later by second pass at right angle to first |
| June and July 1995 | Basal application of oxyfluorofen @ 0.7 kg $ha^{-1}$ |
| August 1995 | One pass disking, followed 2 weeks later by second pass at right angle to first |
| Summer 1995 | Insect control for cottonwood leaf beetles (carbaryl @ 0.92 kg $ha^{-1}$) |
| June 1996 | Insect control for cottonwood leaf beetles (carbaryl @ 0.92 kg $ha^{-1}$) |
| June and July 1996 | One pass disking |
| March 1997 | Plant Nuttall oak seedlings at offset position from cottonwood |
| Winter 2004 | Cottonwood pulpwood harvest |

Four permanent measurement plots were installed in each treatment plot in fall 1995. Measurement plots are rectangular, approximately 0.2 hectare in size for the cottonwood and planted treatments, and 0.06 hectare for the sown treatments. Plot corners were marked with PVC pipe. Measurements for every stem found in each measurement plot included height (from root collar to terminal of main leader), diameter at 15 cm and dbh (for cottonwood only). Height was measured to the nearest cm using a meter tape, and diameters were measured to the nearest 0.01 cm using a 15 cm dial caliper.

Statistical analysis followed a Model I three-treatment ANOVA. The SAS statistical package (SAS Institute, Inc. 1990) was used for data analysis, incorporating Duncan's New Multiple range test for mean comparisons. All significant differences are reported at $\alpha \leq 0.05$.

## RESULTS

Survival after one growing season was significantly different among treatments, and averaged 92% for cottonwood, 63% for planted Nuttall oak seedlings, and 11% for Nuttall oak germinants (Figure 1). No data were collected from the natural succession areas, as a preliminary survey revealed no woody stems present. Number of stems per hectare was significantly different among treatments, with an average of 794 $ha^{-1}$ for the cottonwood, 484 $ha^{-1}$ for the planted seedlings, and 296 $ha^{-1}$ for the germinants (Figure 2). The average total height was significantly greater for the cottonwood (2.1 m) compared to the oak; planted oak seedlings (0.4 m) were taller than germinants (0.1 m), but this difference was not significant (Figure 3). The total average diameter at 15 cm was significantly different among all three treatments, with cottonwood have the largest average diameter, 2.9 cm, followed by the planted seedlings, 0.32 cm, and the germinants, 0.25 cm (Figure 4).

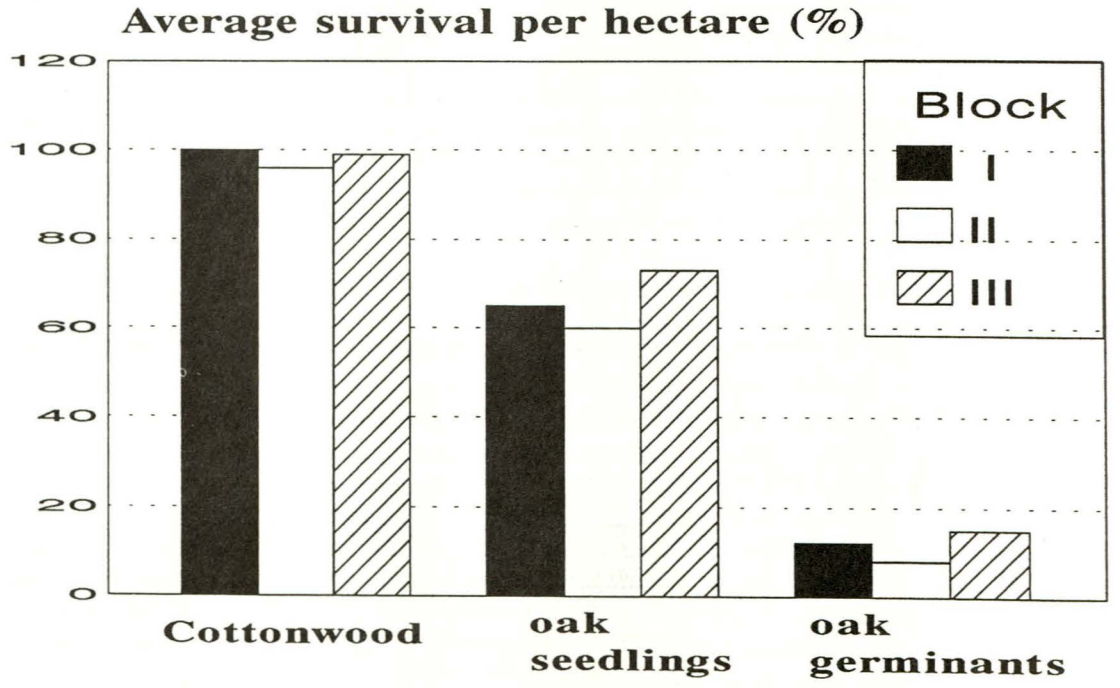


Figure 1. Average survival per hectare, by block, after the first year. Treatments were significantly different at $\alpha \leq 0.05$.

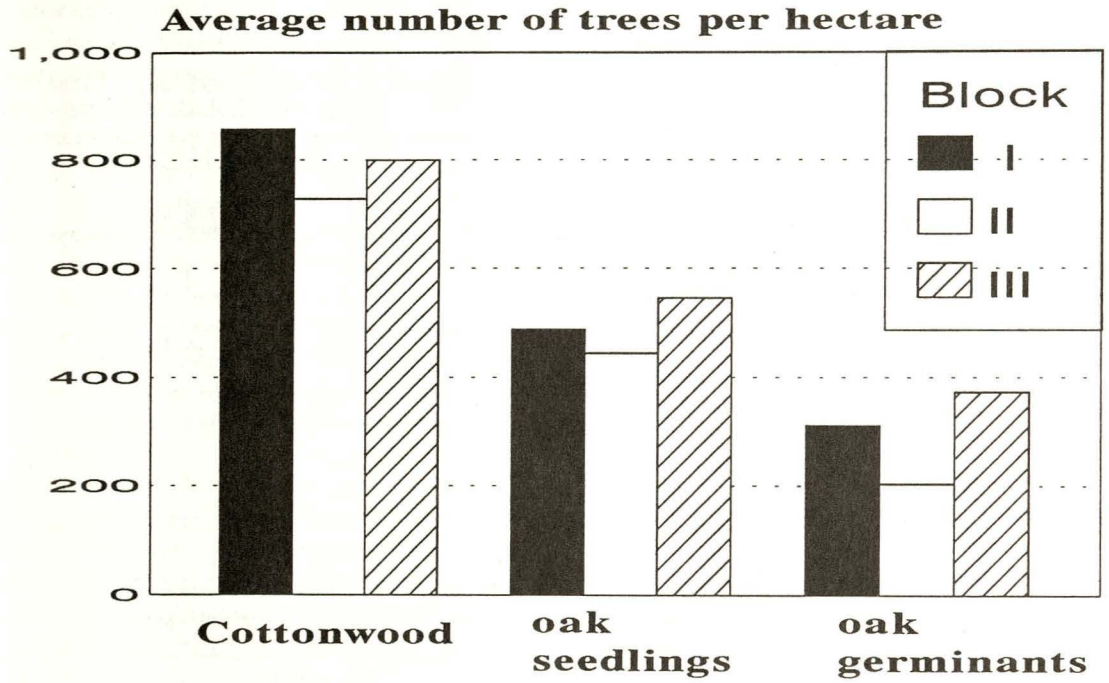


Figure 2. Number of trees per hectare, by block, after the first year. Treatments were significantly different at $\alpha \leq 0.05$.

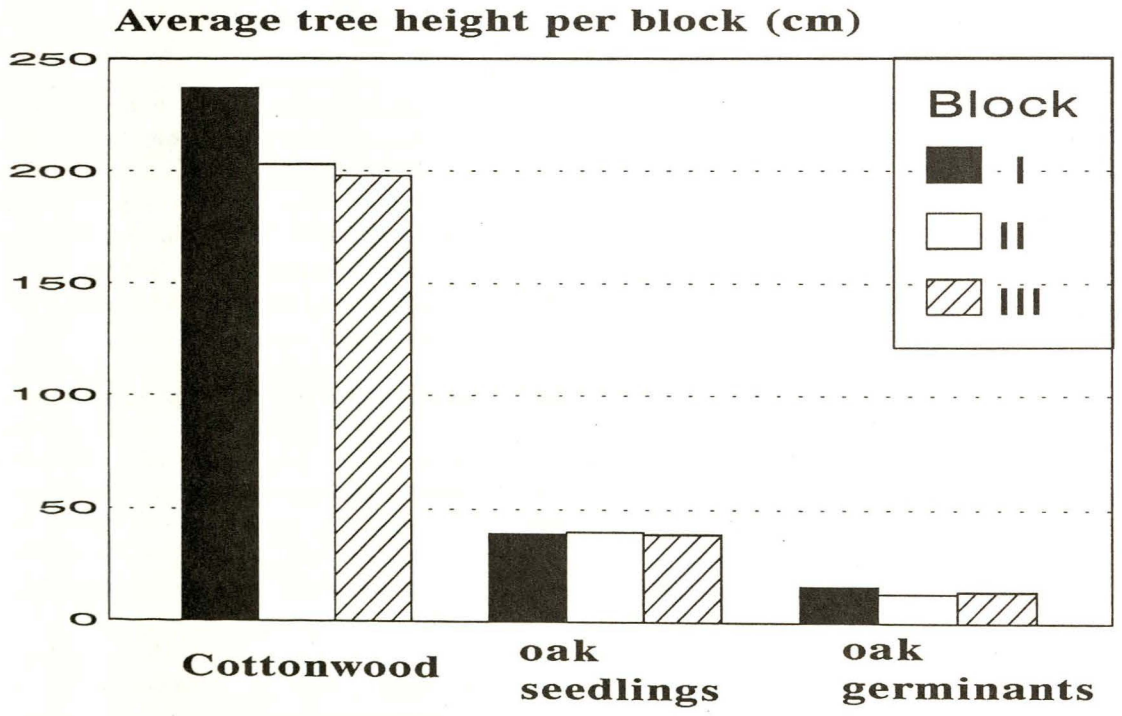


Figure 3. First year average total height per block; cottonwood height was significantly greater than oak seedlings and oak germinants at $\alpha \leq 0.05$.

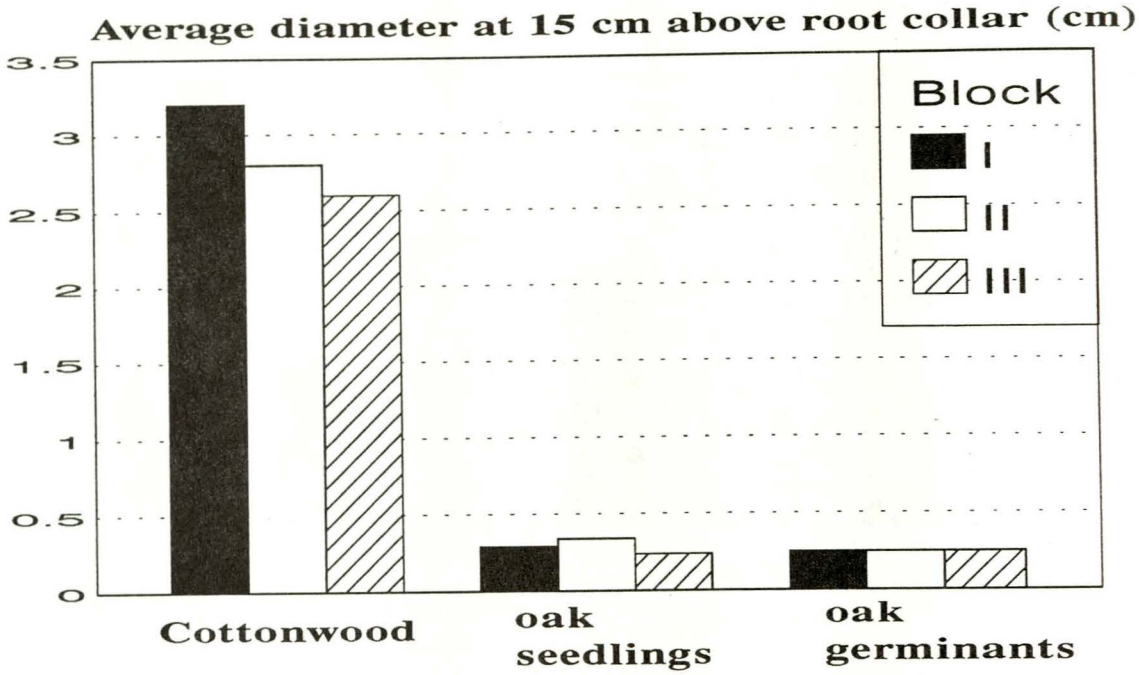


Figure 4. First year total average diameter at 15 cm; treatments were significantly different at $\alpha \leq 0.05$.

## DISCUSSION

Selected tree species were established on an abandoned agricultural field using three different techniques. Results for the first growing season were as expected, although stocking levels in the direct seeded treatment were lower than anticipated. These results support other work over the last 25 years that has shown that bottomland oaks can be established using 1-0 bareroot nursery stock or by direct seeding acorns on sites typical of the lower Mississippi alluvial valley (Allen 1990; Baker and Blackmon 1973; Johnson and Krinard 1985, 1987; Krinard and Kennedy 1987a; Miwa and others 1992). That reforestation can work with little site preparation and no weed control is especially attractive to public agencies, as it allows for the re-establishment of a particular habitat with a minimal commitment of funds and manpower.

Few studies exist, however, where planted seedlings and direct seeding are concurrently compared on the same site. Variability among blocks existed, but the above ground growth performance of planted seedlings and of direct seeded germinants in the establishment year was similar. Direct seeded oaks develop natural root systems on site, thus eliminating the root injury associated with planting bareroot stock. It appears as if the bareroot seedlings in this study spent the establishment year restoring a balance between roots and shoots; however, no root measurements were made. Others have found that the height growth of direct seeded red oak may equal that of 1-0 bareroot seedlings several years after planting (Zaczek and others 1993). Subsequent growth measurements will allow for more definitive results on growth characteristics of seedlings versus germinants.

Survival and stocking levels must be viewed in the context of a landowner's objectives. It should be noted that the extremely high cottonwood survival percentage was enhanced by ideal planting conditions and the experience of Crown Vantage's planting crew. For the cottonwood plantation, the primary objective is fiber production, and the high survival and increases in growth justify weed control costs necessary to achieve full stocking. In a short time period, these cottonwood plantations create a visual habitat that is also desired by many landowners. If enhancement of the hard mast component in stands established to create wildlife habitat is the objective, weed control may not be needed.

Stocking levels obtained by planting seedlings were generally higher than those obtained by direct seeding, but these data reflect only one growing season and may change because of delayed germination of the acorns. Delayed germination over the 2 years following planting may increase the stocking rates. Miwa and others (1993) found a slow, steady germination of Nuttall oak throughout the growing season on similar soils. We have observed delayed germination for oak acorns on nearby reforested land under the management of the Fish & Wildlife Service. The expected survival of operationally direct seeded acorns has been reported at 35% (Kennedy 1993, Krinard and Johnson 1985). The greater number of planting spots per ha for direct seeding (3073 $ha^{-1}$ as compared to 714 $ha^{-1}$ for bare-root seedlings) is designed to compensate for expected differences in survival. When hardwoods are planted without weed control on cleared clay soils, some species are better able to survive and compete than others. It has been noted that the slow growth of oaks in the first 5 years may be discouraging because of competition outgrowing the oaks. However, Wittwer (1991) found that competition control did not improve regeneration success of direct seeded oaks on a bottomland site in southeastern Oklahoma, and Krinard and Kennedy (1987b) found that Nuttall oak survival, without weed control, was 80% after 16 years. Research by Johnson (1981) showed that Nuttall oak in particular is a late grower that becomes increasingly more competitive as it persists and develops in the reproductive stand.

Tree planting on land enrolled in the Wetland Reserve Program (WRP) must produce 309 desirable woody stems per hectare after three growing seasons. According to these guidelines, stocking of cottonwood and planted oak seedlings is acceptable in this study. Unless delayed germination occurs, stocking in the direct seeded treatment does not meet WRP minimal stocking levels. Variable success with direct seeding has caused some states to require seedling planting.

Allen (1990) suggested several reasons why stocking and survival of direct seeded stands might be lower than those planted with seedlings, including use of unsuitable genetic stock or lower viability, rodent predation, and inability of very small seedlings to withstand drought and/or weed competition. Local seed collection and float testing of acorns ensured that quality, viable seed was used in this study. A pilot small mammal trapping study (600 trap-nights) conducted on a single plot of the direct seeded and the cottonwood treatments in October 1995 indicated that very dense small mammal populations were present on the direct seeded treatment (202 animals captured 216 times) compared to the cottonwood (2 animals captured 2 times). *Sigmodon hispidus* was the most abundant of five species captured. (Hamel and others, in review). Therefore, rodent predation of acorns may have contributed to the lower survival rates and stocking in the direct seeded treatments, which had extensive weed cover compared to the bare soil conditions in the cottonwood treatments. Alternatively, soil cracking may have exposed new germinants to desiccation.

Cottonwood height and diameter growth greatly exceeded that of the oak (as would be expected by the fastest growing tree native to North America) (Capel and Coffman 1966). Growth was better for the planted oak seedlings compared to the germinants, with planted seedling height being four times greater than germinant height. Although there was much variability in height growth, the difference between planted and sown oaks was not statistically significant.

## SUMMARY AND CONCLUSION

Planting oak seedlings and direct seeding acorns were used to establish an oak component on an abandoned agricultural field. First year performance results indicated better survival rates and stocking for planted seedlings compared to oak germinants. However, height growth of surviving oaks did not differ between seedlings and germinants. The intense cultivation involved in establishing cottonwood was manifested by high survival rates and stocking.

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# REGENERATION RESPONSES OF OAK-DOMINATED STANDS TO THINNING AND CLEARCUTTING IN NORTHWESTERN PENNSYLVANIA

John A. Stanturf [1], L. R. Auchmoody [2], and Russell S. Walters[2]

Abstract: Regenerating mature oak stands on better sites is difficult on the Allegheny Plateau because of abundant advance regeneration of faster growing competitors. We have observed that oak stands on the Allegheny National Forest (ANF) in northwestern Pennsylvania almost always regenerate to cherry-maple rather than to oak-hickory stands. Such conversion greatly concerns land managers and the general public. We surveyed 30 stands, varying in size from 8 to 42 acres, in a 786 acre compartment on the Sheffield Ranger District to document typical responses of oak-hickory stands on good sites to partial and complete overstory removal. These stands were 90 to 110 years-old when cut. Five stands were clearcuts that originated from previously uncut stands; 18 stands had been partially cut; and 7 stands remained uncut. The five clearcuts were 4 years old (1 stand), 7 years-old (1), and 15 years old (3) when examined. The 7 year old stand was fertilized 2 years after clearcutting. The 18 thinned stands were treated 6 years before we examined them. Trees to be cut were individually marked by ANF staff and commercially harvested. Advance reproduction in thinned stands varied from 0% to 75% of regeneration plots stocked with preferred species. Black cherry was more abundant in the advance reproduction than in the overstory. Advance reproduction of sapling size was present in 15 stands, occurring in up to 40% of plots in a stand. Most saplings were black cherry or red maple. Black cherry was the most numerous species in the clearcut stands and the only species consistently in the dominant position of the new age class. Isolated patches of dominant and co-dominant oak saplings were in the main canopy of the new age class on dry microsites and along skid trails. Oak seedlings were present in the new age class, but all were less than 1 ft tall. The unfertilized clearcut stands grew at essentially the same rate, or considerably faster than previously sampled cherry-maple stands. Response to fertilizer was of a magnitude similar to that in previously sampled cherry-maple stands.

## INTRODUCTION

Regeneration greatly concerns forest managers throughout portions of the US with abundant amounts of oak community types (Clark and Watt 1971, Fosbroke and Carvel 1989, Holt and Fisher 1979, Loftis and McGee 1993, Weitzman and Trimble 1957). Oak regeneration is particularly difficult to obtain on better sites where advance regeneration of faster growing competitors is often abundant (Abrams and Downs 1990, Abrams and Nowacki 1992, Heiligmann and others 1985). Besides losing the high value red and white oak for commodity uses, the conversion of predominantly oak stands to other hardwood types concerns land managers because of the loss of biodiversity at the community level (McMinn 1991) and hard-mast for wildlife.

Oak stands on the Allegheny National Forest (ANF) in northwestern Pennsylvania are similar in many respects to stands throughout the Allegheny Plateau of New York, Pennsylvania, Ohio, and West Virginia. Regeneration of commercial species, including oaks, in the ANF is complicated by high populations of whitetail deer (Tilghman 1989). Aerial fertilization of recent clearcuts is common on the ANF, to overcome nitrogen and phosphorus deficiencies that lead to poor height development (Auchmoody 1982) that exposes the seedlings longer to deer browsing. This can result in a regeneration failure (Marquis 1974).

We have observed that oak stands on the ANF almost always regenerate to cherry-maple rather than to oak-hickory stands. Such conversion greatly concerns managers and the public (FEIS 1986). Deferring treatments until proven

[1] Project Leader, Center for Bottomland Hardwood Research, Southern Research Station, USDA Forest Service, P.O. Box 227, Stoneville, MS 38776.

[2] Principal Silviculturists, now retired, Northeastern Forest Experiment Station, P.O. Box 928, Warren, PA 16365.

methods are available for regenerating stands to oaks is not a viable option in the face of gypsy moth and drought-induced mortality. We undertook this survey of one compartment typical of oak-hickory and transition hardwood stands on the Allegheny National Forest to document typical responses of these stands to partial and complete overstory removal. Although not a designed study, our approach illustrates the principle of adaptive management, where every management action can be viewed as an experiment.

## METHODS

### Study Area

All stands were in Compartment 4 of the Sheffield Ranger District, Allegheny National Forest (ANF). This compartment of 786 acres is within two miles of the Allegheny River. Climate is moderate, with cold, snowy winters and mild, warm summers. The average winter temperature is 28°F, and summer temperatures average 60°F. Total precipitation is 43 inches; about one-half falls as rain between April and December. Average seasonal snowfall is 74 inches. Dominant soils belong to the Hazleton (loamy-skeletal, mixed, mesic Typic Dystrochrepts), Cookport (fine loamy, mixed, mesic, Aquic Fragiudults), Cavode (clayey, mixed, mesic Aeric Ochraquults), and Gilpin (fine loamy, mixed, mesic Typic Hapludults) series which are productive for upland oaks, with northern red oak (*Quercus rubra* L.) site indices of 70 ft to 83 ft (Cerutti and others 1985).

Thirty stands, each covering 8 to 42 acres, were sampled: 5 clearcuts (Stands CC1-CC4, CCF5) that originated from previously uncut stands, 18 thinned stands, and 7 uncut stands. Of the five clearcuts, Stands CC1-CC3 were harvested in 1970; Stand CCF5 in 1978; and Stand CC4 in 1980. The stands were 15, 7, and 4 years old when examined. Only Stand CCF5 was fertilized in 1980 (2 years after cutting). The 4 year-old, Stand CC4, was approximately 100-110 years old at the time of overstory removal, based on stump ages of three dominant northern red oak.

The 18 thinned stands were treated in 1979, 6 years before we examined them. Trees to be cut were individually marked by ANF staff and stands were commercially harvested. Trees per acre varied widely following thinning (Table 1), from a low of 91 in Stand N5 (north aspect) to a high of 378 in Stand S2 (south aspect). Average stand diameters of 13.4 inches to 17.7 inches (approximately equal to median diameter; see Marquis and others 1992 for justification). The low correlation between basal area and relative density ($r^2$= .42) reflects the mixed species composition as well as differences due to thinning. In two-thirds of the stands, relative density was within the recommended 60% to 80% for sawtimber management (Marquis and others 1992). In the remainder of the thinned stands, relative density was 46% to 58%.

All but five of the thinned stands were classified as oak-hickory (Table 1), based on residual stand composition. The five northern hardwood stands all had an oak component that ranged from 8% to 24% of basal area (Table 2). Northern red oak and red maple (*Acer rubrum* L.) were the only species common to the overstory of all residual stands (Table 2). Black cherry (*Prunus serotina* Ehrh.) was a minor component in most stands. Other common associates included yellow-poplar (*Liriodendron tulipifera* L.) and cucumber tree (*Magnolia acuminata* L.).

The seven uncut stands resembled the thinned stands in most respects. Relative density ranged from a high of 93% to a low of 47% (Table 1). Average stand diameters ranged from 12.8 inches to 17.4 inches. One stand (N1) was predominantly red maple (60% of basal area) but had an oak component (7%). The other stands ranged from 34% to 79% oak basal area.

### Measurements

Standard USDA Forest Service stand examination procedures were used in all stands except recent clearcuts (Marquis and others 1984). These included 10 baf prism plots for overstory trees (1.0 inch or larger dbh) with dbh tallied by 2-inch classes. Regeneration tallies were made on 20 or more 6-ft radius plots in each stand, including stems 5.50 inches or smaller dbh. Aspect was simplified into cardinal directions from compass readings: East=45° to 134°, South=135° to 224°, West=225° to 314°, and North=315° to 44°.

Table 1. Thinned and uncut stands in Compartment 4, Sheffield Ranger District, Allegheny National Forest in 1985 were mostly oak-hickory stands

| Stand number | Aspect | Stand type[1] | Age[2] | Trees acre$^{-1}$ | Stand diameter[3] | Basal area | Relative density | Sapling[4] | Oak sprout potential |
|---|---|---|---|---|---|---|---|---|---|
| **Thinned** | | | | | Inches | ft$^2$ | - - - Percent - - - | | Trees acre$^{-1}$ |
| E1 | E | O | 95 | 173 | 17.4 | 76 | 46 | 40 | 7 |
| E2 | E | N | 93 | 200 | 16.5 | 84 | 54 | 17 | 2 |
| E3 | E | O | 99 | 138 | 16.3 | 78 | 60 | 20 | 5 |
| E4 | E | O | 93 | 126 | 15.8 | 75 | 54 | 5 | 13 |
| E7 | E | N | 76 | 334 | 13.4 | 120 | 78 | 0 | 18 |
| N3 | N | O | 91 | 221 | 14.5 | 74 | 65 | 25 | 5 |
| N4 | N | O | 88 | 254 | 15.1 | 91 | 67 | 25 | 8 |
| N5 | N | O | 94 | 91 | 17.7 | 90 | 54 | 15 | 6 |
| S1 | S | N | 89 | 362 | 14.8 | 96 | 70 | 20 | 7 |
| S2 | S | O | 97 | 378 | 14.5 | 88 | 80 | 5 | 9 |
| S3 | S | O | 93 | 148 | 16.5 | 82 | 58 | 17 | 7 |
| S4 | S | O | 92 | 222 | 16.0 | 87 | 64 | 10 | 15 |
| S5 | S | O | 87 | 163 | 15.7 | 84 | 55 | 0 | 9 |
| W1 | W | N | 97 | 292 | 17.2 | 109 | 68 | 36 | 1 |
| W2 | W | N | 93 | 147 | 16.9 | 106 | 60 | 5 | 1 |
| W5 | W | O | 82 | 212 | 14.1 | 92 | 68 | 0 | 21 |
| W7 | W | O | 89 | 279 | 14.0 | 89 | 75 | 36 | 12 |
| W8 | W | O | 89 | 183 | 15.2 | 83 | 61 | 35 | 2 |
| **Uncut** | | | | | | | | | |
| E5 | E | O | 74 | 403 | 13.0 | 113 | 76 | 20 | 31 |
| E6 | E | O | 96 | 165 | 17.4 | 83 | 47 | 20 | 4 |
| N1 | N | N | 97 | 430 | 17.0 | 102 | 65 | 29 | 1 |
| N2 | N | O | 90 | 220 | 16.2 | 106 | 72 | 50 | 7 |
| W3 | W | O | 87 | 358 | 13.9 | 118 | 93 | 14 | 35 |
| W4 | W | O | 88 | 199 | 15.5 | 92 | 68 | 14 | 15 |
| W6 | W | O | 87 | 380 | 12.8 | 83 | 74 | 27 | 36 |

[1] Stand type: O=Oak-Hickory, N=Northern Hardwoods.
[2] Effective age.
[3] Stand diameter is approxiamtely the media stand diameter.
[4] Saplings are the percentage of regeneration plots stocked with saplings of preferred species.

The SILVAH prescription software (Marquis and others 1992) was used to summarize these data and calculate stems per acre, basal area, volume, average stand diameter, and relative density. Relative stand density is the ratio of absolute stand density (taken as basal area) to a reference level of average maximum density of an undisturbed stand of similar species composition and diameter distribution (Marquis and others 1992). We also used SILVAH to determine regeneration adequacy from the understory data and stand type from overstory dominance calculated from basal areas of residual stands.

Table 2. Species composition of the overstory in thinned (residual) and uncut stands

| Stand number | Basal area | Oak | Red oak[1] | White oak | Black oak[2] | Red maple | Black cherry |
|---|---|---|---|---|---|---|---|
| | $ft^2$ | Percent | | | | | |
| **Thinned** | | | | | | | |
| E1 | 76 | 53 | 45 | 7 | 1 | 33 | 5 |
| E2 | 84 | 24 | 16 | 8 | 0 | 40 | 9 |
| E3 | 78 | 59 | 14 | 44 | 1 | 14 | 0 |
| E4 | 75 | 74 | 23 | 39 | 12 | 12 | 3 |
| E7 | 120 | 22 | 22 | 0 | 0 | 20 | 15 |
| N3 | 74 | 63 | 7 | 55 | 1 | 26 | 3 |
| N4 | 91 | 40 | 9 | 31 | 0 | 30 | 13 |
| N5 | 90 | 40 | 31 | 8 | 1 | 41 | 0 |
| S1 | 96 | 24 | 14 | 9 | 1 | 19 | 11 |
| S2 | 88 | 73 | 7 | 60 | 6 | 22 | 2 |
| S3 | 82 | 61 | 34 | 21 | 6 | 28 | 2 |
| S4 | 87 | 73 | 43 | 23 | 7 | 21 | 1 |
| S5 | 84 | 52 | 36 | 14 | 2 | 29 | 0 |
| W1 | 109 | 12 | 8 | 4 | 0 | 59 | 3 |
| W2 | 106 | 8 | 8 | 0 | 0 | 47 | 10 |
| W5 | 92 | 79 | 40 | 26 | 10 | 19 | 2 |
| W7 | 89 | 56 | 14 | 37 | 5 | 12 | 0 |
| W8 | 83 | 36 | 5 | 28 | 3 | 40 | 5 |
| **Uncut** | | | | | | | |
| E5 | 113 | 49 | 42 | 7 | 0 | 24 | 9 |
| E6 | 83 | 34 | 33 | 1 | 0 | 41 | 5 |
| N1 | 102 | 7 | 7 | 0 | 0 | 60 | 10 |
| N2 | 106 | 36 | 11 | 25 | 0 | 38 | 17 |
| W3 | 118 | 79 | 36 | 25 | 18 | 10 | 2 |
| W4 | 92 | 65 | 38 | 15 | 12 | 26 | 0 |
| W6 | 83 | 77 | 18 | 48 | 11 | 9 | 0 |

Regeneration adequacy was evaluated according to procedures detailed in Marquis and others (1984) and Sander and others (1976) for black cherry, northern hardwoods, large and small oak seedlings, and saplings. SILVAH was used to calculate oak sprouting potential using the procedure of Sander and others (1976) from overstory data on species and dbh. This estimate of sprouting potential was reduced to account for heavy deer pressure (Marquis and others 1984).

We examined the clearcuts using 0.01 acre circular plots. Three of the stands (CC1, CC2 CC3) were 15 years old when examined and we tallied the overstory by species and dominance class. We estimated annual height growth with a height pole on 20 dominant black cherry stems in each of three stands: a 15 year old clearcut (Stand CC2), a 7 year old fertilized clearcut (Stand CCF5), and a 4 year old unfertilized clearcut (Stand CC4).

Staff of the ANF surveyed stocking in Stands CC4 and CCF5 the summer after harvest and 2 years later. Seedling regeneration on 20 or more 6-ft radius plots was assessed in each stand. They differentiated seedlings into three height classes: taller than 5-ft, 3-ft to 5-ft, and under 3-ft. Preferred species (black cherry, red maple, white ash, oaks, yellow poplar, basswood (*Tillia americana* L.), cucumber tree, and hemlock) were tallied separately from non-preferred (beech (*Fagus grandifolia* L.) and the birches). The percentage of black cherry in the preferred class was estimated. The plots were evaluated for adequacy of stocking by preferred species according to the following criteria: Class I, 2 or more seedlings 5-ft or taller; Class II, 5 or more seedlings at least 3-ft tall; and Class III, 25 or more seedlings of any height. Any plot meeting one of these criteria was judged to be adequately stocked.

## RESULTS

### Regeneration in Partially Cut and Uncut Stands

From none to 75% of regeneration plots within the 18 thinned stands were stocked with preferred species (Figure 1). Black cherry was more prevalent in the new age class than in the overstory. Besides red maple, the other preferred regeneration was predominantly white ash and yellow-poplar. Oak regeneration was entirely small oak (less than 4.5 ft tall) in five of the six stands where it occurred (Figure 1). Species included northern red oak, white oak (*Q. alba* L.) and black oak (*Q. velutina* Lam.).

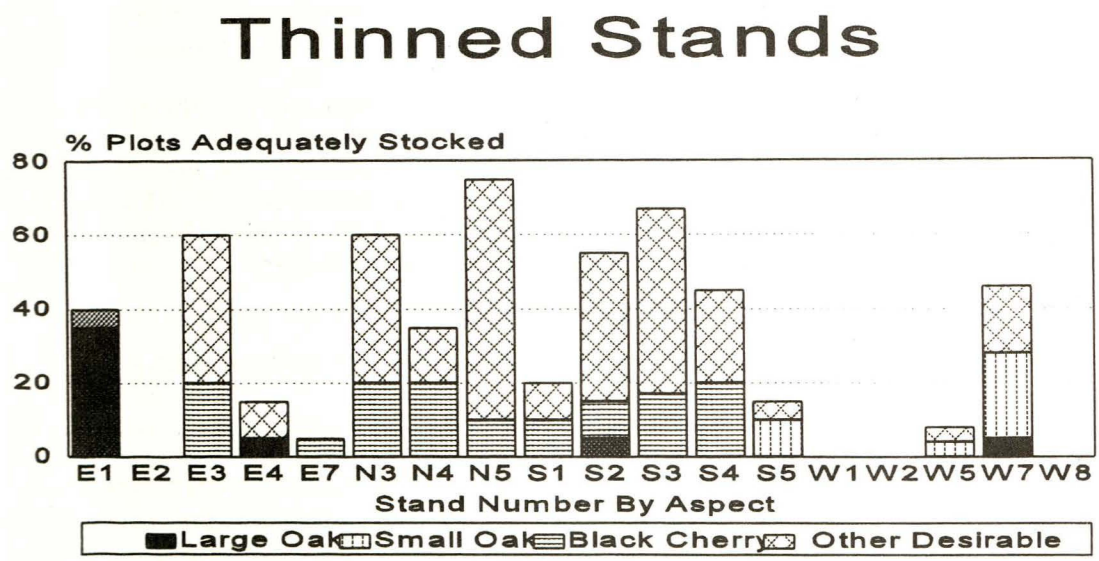


Figure 1. Percentage of the regeneration plots in thinned stands that were adequately stocked with oak (large or small), black cherry, or northern hardwoods (includes white ash, red or sugar maple, yellow-poplar, basswood, cucumber tree, or eastern hemlock).

Saplings were present in all but three stands, with up to 40% of plots having stems of this size (Table 1). Most saplings were black cherry or red maple. Other species were yellow birch (*Betula alleganiensis* Britton) and sweet birch (*Betula lenta* L.), hop-hornbeam (*Ostrya virginiana* (Mill.) K. Koch ), blue beech (*Carpinus caroliniana* Walt.), red oak, white pine (*Pinus strobus* L.), cucumber tree and eastern hemlock (*Tsuga canadensis* L.). Sprouting potential (Sander and others 1976) was as high as 21 oak trees per acre, but most stands had fewer than 10 oak trees per acre likely to produce acceptable sprouts (Table 1). For the range of oak diameters in these stands, at least 46 trees per acre are needed to add 5% to regeneration stocking under heavy deer pressure (Marquis and others 1984).

From 5% to 50% of regeneration plots in the uncut stands were stocked with preferred species (Figure 2). Stand W3 had the highest relative density of any stand we examined (93%) and high regeneration stocking; 50% of the plots were adequately stocked. This uncut stand was also one of the few stands with oak regeneration.

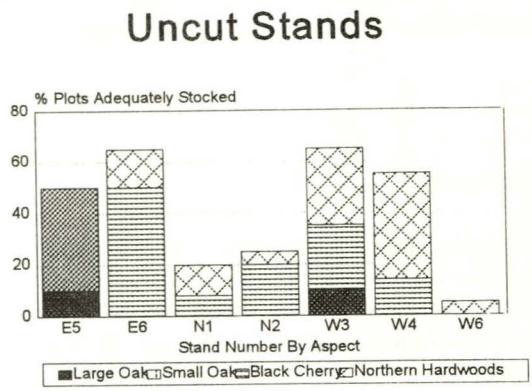


Figure 2. Percentage of the regeneration plots in uncut stands adequately stocked with oak, black cherry, or other desirable species (designations as in Figure 1).

Plots adequately stocked with oak regeneration occurred only in stands where more than 50% of basal area was oak (Table 3). This was true for both uncut and thinned stands. It was equally likely, however, that a stand with greater than a 50% oak basal area would have no oak regeneration. The thinned stands with oak regeneration ranged from 52% to 76% oak basal area, and the thinned stands greater than 50% oak basal area without oak regeneration ranged from 53% to 73% oak basal area. Thus, the probability of obtaining some oak component in the regeneration was 50% in thinned stands with greater than 50% oak basal area in the residual overstory.

Table 3. Relationship of oak advance regeneration to percentage of overstory basal area as oak species

| | Some oak regeneration | No oak regeneration | Total |
|---|---|---|---|
| Oak BA> 50% | Thinned: 5<br>Uncut: 1 | Thinned: 5<br>Uncut: 2 | Thinned: 10<br>Uncut: 3 |
| Oak BA< 50% | Thinned: 0<br>Uncut: 0 | Thinned: 8<br>Uncut: 4 | Thinned: 8<br>Uncut: 4 |
| Total | Thinned: 5<br>Uncut: 1 | Thinned: 13<br>Uncut: 6 | Thinned: 18<br>Uncut: 7 |

Clearcut Stands

Stands CC1-CC3 show the range of species composition that results when previously uncut stands are clearcut (Table 4). Black cherry was the most numerous, and the only species in the dominant position of the new age class in all stands. Red and white oak and other hardwoods were, however, among the dominants and codominants in Stand CC3.

Table 4. Species composition by dominance class 15-years after clearcut harvest, stems per acre

| | Stand CC1 | | | |
|---|---|---|---|---|
| | Dominant | Co-dominant | Intermediate | Overtopped |
| Black cherry | 225 | 50 | 75 | 450 |
| Red maple | | | 25 | 325 |
| White ash | 25 | 50 | 25 | 333 |
| Red oak | | | | 75 |
| Hop-hornbeam | | 25 | 25 | 75 |
| Sweet birch | 50 | 225 | 50 | 125 |
| Sugar maple | 25 | | | |
| Ironwood | | | | 100 |
| *Hamamelis virginiana* (L.) | | | | 75 |

| | Stand CC2 | | | |
|---|---|---|---|---|
| | Dominant | Co-dominant | Intermediate | Overtopped |
| Black cherry | 300 | 250 | 150 | 500 |
| Red maple | | 50 | 25 | 650 |
| White ash | | | | 50 |
| Beech | | | | 50 |
| *Hamamelis virginiana* (L.) | | | | 25 |

| | Stand CC3 | | | |
|---|---|---|---|---|
| | Dominant | Co-dominant | Intermediate | Overtopped |
| Black cherry | 100 | 125 | 150 | 200 |
| Red maple | 75 | 300 | 50 | 850 |
| Red oak | 50 | 150 | 25 | 575 |
| White oak | | 25 | | 125 |
| *Populus tremuloides* Michx. | 25 | | | 75 |
| *Prunus pennsylvanica* L.f. | | 25 | | |
| *Amelanchier* spp. | | | | 100 |
| *Aralia* spp. | | 25 | | |

None of the clearcut stands regenerated to oak. We did find isolated patches of dominant and codominant oak stems in the main canopy of the new age class on dry microsites, and adjacent to skid trails where competition from other species was absent around at least 50% of the crown circumference (data not presented). This edge effect favoring oak regeneration has been noted by others (Whitney and Runkle 1981, Crow 1988). Chestnut (*Castanea dentata* (Marsh.) Borkh.) persisted in openings for a number of years after logging. Oak seedlings were present in the new age class, but all were less than a foot tall. Understory plants typical of oak stands (e.g., *Rhododendron* spp., *Kalmia latifolia* (L.), *Sassafras albidum* (Nutt.) Nees, hop-hornbeam, *Vaccinium* spp., and *Cornus florida* (L.)) were reduced or absent in the 15-year-old stands (Table 4). Stocking surveys made in Stands CC4 and CCF5 the season following completion of cutting and again 2 years illustrate the early development of regeneration following clearcutting. Stand CC4 was well-stocked with regeneration at the time of the first post-harvest survey (Figure 3). Black cherry comprised 67% of the tallest seedlings (over 5-ft), 76% of the seedlings 3-ft to 5-ft tall, and fully 90% of the smaller seedlings. Two years later, 100% of the plots were stocked with seedlings taller than 5-ft. Stand CCF5 had lower initial stocking in the larger size classes and was fertilized to stimulate seedling growth. Height growth was accelerated such that the number of seedlings 3-ft and taller tripled. The percentage of black cherry stocking declined between the two surveys and red maple increased; overtopped oak seedlings were also present.

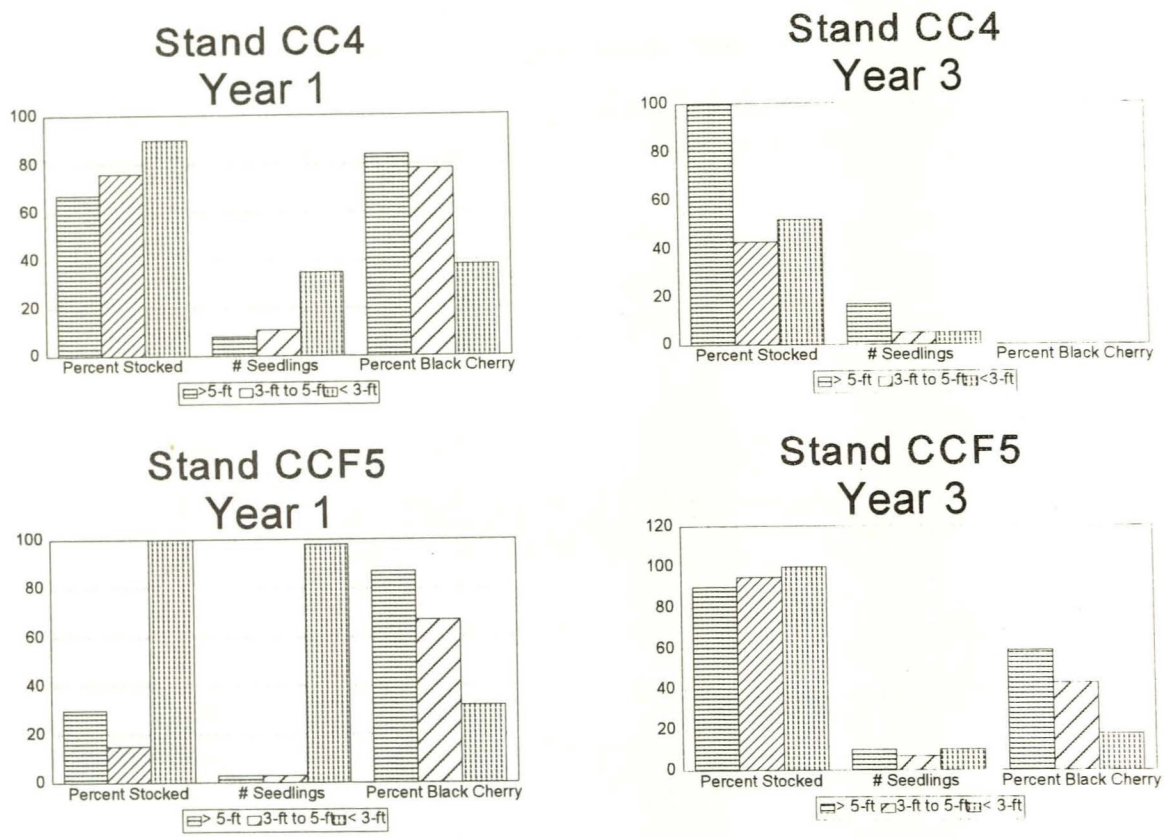


Figure 3. Regeneration stocking (percent of plots adequately stocked, number of seedlings per plot, and percentage black cherry stems) in Stand CC4 (unfertilized) and Stand CCF5 (fertilized) 1 and 3 years aftrer harvest. Seedlings were tallied in three height classes: > 5 ft, 3-ft to 5-ft, and < 3-ft.

The pattern of cumulative height of dominant black cherry in Stands CC2, CC4, and CCF5 (this study) is compared in Figure 4 to data from unfertilized (CM1) and fertilized (CM2) stands that developed after clearcutting cherry-maple stands. Black cherry in unfertilized Stands CC2 and CC4 grew at the same rate, or somewhat faster than in CM1, the unfertilized cherry-maple stand ( Auchmoody 1982). Response to fertilizer in Stand CCF5 was of a magnitude similar to that in a previously reported cherry-maple stand, CM2 (Figure 4). We could not test the fertilizer response statistically as no true control existed.

## DISCUSSION

The regeneration of valuable oak-hickory and cherry-maple forests in northwestern Pennsylvania has failed in the past due to myriad problems, including browsing by deer (Marquis 1974, Marquis and Brenneman 1981), interfering plants (Horsley and Marquis 1983), nutrient deficiencies (Auchmoody 1982), and insect predation (Galford and others 1991). All these factors contribute to insufficiency of advance regeneration at the time of overstory removal and a failure of surviving seedlings after release. Stands in the study area were typical of oak-hickory and transition hardwoods found on good growing sites. Red oak site index estimates for the predominant soils were all above the SI 70 threshold set by Sander and others (1976) as the upper limit for use of their regeneration guidelines. Aspect had no apparent influence on overstory or understory species composition, contrary to other studies (Auchmoody and Smith 1979, Trimble and Weitzman 1956). Evenly distributed and ample growing season precipitation, frequent cloud cover, and low summer temperatures resulting in general lack of soil moisture deficits probably counteracted effects of aspect noted elsewhere.

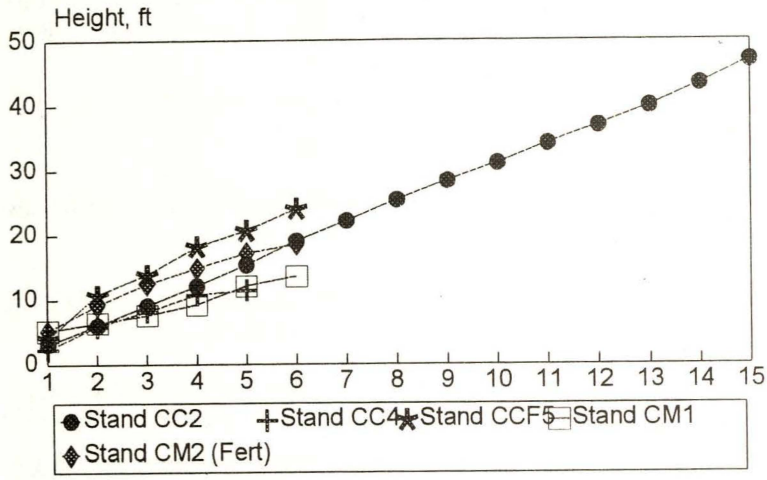


Figure 4. Cumulative height growth in clearcut stands regenerated from previously uncut oak-hickory (Stand CC2, CC4 and CCF5) or cherry-maple overstory (Stand CM1 CM2). Data for cherry-maple stands from Auchmoody (1982). Stands CCF5 and CM2 were fertilized 2 years after clearcutting.

The seven uncut stands represent the range of overstory and understory species composition found in the study area. Except for one stand classed as northern hardwoods (N1), these stands were dominated in the overstory by oaks or had a sizeable oak component. Red maple was the other dominant species. Black cherry occurred in the overstory, but was a minor component (0% to 10%). In the understory, however, oak seedlings occurred in only one stand (W4). We believe these uncut stands typify the conditions of the other stands in the compartment before thinning or clearcutting. Unfortunately, no records exist for pre-thinning regeneration tallies.

Thinning was the intent of the cuttings, however they can be viewed as establishment cuttings in a shelterwood sequence. Even stands dominated by oaks did not develop many oak seedlings consistently when thinned to the target 60% to 80% relative density. From these stands it appears, however, that at least 50% of residual stand basal area in oaks was a necessary, but not sufficient, condition for oak regeneration to develop. Abundant regeneration of other preferred species developed in most stands following thinning, although a third of the stands had less than 10% stocked plots.

The loss of oak types is cause for concern about loss of community diversity (McMinn 1991) and regulations require land managers to maintain an oak component on 20% of the Allegheny National Forest (FEIS 1986). Certainly the dominance of black cherry, at times in virtual single species stands, raises concerns about the risk of catastrophic pest or disease outbreaks. The prevalence of black cherry is due primarily to its prolific seeding and the preference for other species by deer (Hough 1965, Marquis 1975, Whitney 1984). The ability of black cherry to seed into stands with little or no presence in the overstory is well illustrated by these data.

When oak stands are mature but oak advance regeneration is lacking, conversion to cherry-maple stands is possible. While the fertilization results presented here are not conclusive, it appears that untreated black cherry on sites previously dominated by oak-hickory stands grow somewhat more vigorously than unfertilized black cherry growing on sites previously dominated by cherry-maple stands. The response to fertilization of Stand CCF5 shows, however, that these stands are somewhat deficient. It appears from these clearcuts that black cherry advance regeneration will grow beyond the reach of deer quickly enough to form a large proportion of the new age class. We doubt that mesic sites such as these can be naturally regenerated to oak, although a stronger foundation of research than these data is needed to be conclusive. Our data agree with other assessments, that oak will likely dominate only the poor sites after clearcutting (Abrams and Downs 1990, Hilt 1985). Further research is needed to develop

effective methods of regenerating oak if biodiversity goals are to be met, probably relying on artificial regeneration. Continuing to regenerate these stands using conventional shelterwood cuttings will regenerate mostly black cherry and red maple, and occasionally some oaks.

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# SURVIVAL AND GROWTH OF HARDWOOD SEEDLINGS FOLLOWING PREPLANTING-ROOT TREATMENTS AND TREESHELTERS

Felix Ponder, Jr.[1]

Abstract: The study evaluated the influence of root collar diameter, number of large lateral roots, preplanting-root treatments (biostimulant called Roots and a moisture loss retardant called supersorb) and tree shelters on 1-0 black walnut (*Juglans nigra* L.) and northern red oak (*Quercus rubra* L.) seedlings. Four years after outplanting, the survival of black walnut and northern red oak increased with the number of large lateral roots. The difference was significant ($p < 0.05$) for black walnut but not for northern red oak. Total seedling height was significantly affect by tree shelters. The height of black walnut and northern red oak was significantly better with tree shelters than without tree shelters. There were significant interactions between the number of large lateral roots and root treatments for height and diameter for both species and between root collar diameter and number of large lateral roots for the diameter of northern red oak.

## INTRODUCTION

Regenerating many desirable hardwoods after harvesting continues to be a problem on many sites in the Central Hardwoods Region. Planting nursery stock on sites that lack adequate advance regeneration, is a possible solution (Johnson 1979). Planted hardwoods often have variable growth rates which can be attributed to properties of the root system (Farmer 1975, Kormanik 1988).

Foster and Farmer (1970) suggested that the use of larger planting stock would help solve the slow growth problem after plantation establishment. Later, Johnson (1976) reported that planting seedlings with stem caliper of 8 mm or greater also increased survival and early growth. In one study, lifted oak seedlings possessed the characteristics to potentially become high-yielding trees 5 years after planting (Johnson 1984). Stroempl (1985) graded 2-0 northern red oak nursery stock from Ontario nurseries for 6 years and found 40% of the seedlings to be of unacceptable quality.

Nurseries are improving seedling quality by increasing the number of large lateral roots, also called first order lateral roots (>1 mm (0.04 in.). The number of large lateral roots are a morphological root characteristic of 1-0 seedlings that remains stable over a range of fertility levels and strongly influences field performance (Kormanik 1985, Kormanik 1986). Johnson (1988) reported that red oak seedlings undercut in the nursery bed had more large lateral roots when lifted and these seedlings survived and grew better when outplanted.

The ability of bare-root seedlings to produce new roots after planting has also been shown to be a valuable index of seedling quality (Ritchie and Dunlap 1980, Stone 1955). The organic biostimulant, "Roots," has been shown to greatly increase root and shoot growth of plants, and to increase seedling resistance to stresses such as low soil water potential and low soil nutrient content (Berlyn and Beck 1980, Berlyn and Russo 1990). Several other products also appear to increase the ability of the soil to hold more water as a means to reduce or moderate moisture stress for planted seedlings. One of these products, SuperSorb-F (supersorb), distributed by Aquatrols Corporation of America in Pennsauken, NJ, was used in this study. Minimizing the stresses encountered by newly planted seedlings could maximize root development over a range of site conditions.

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[1]Supervisory Research Soil Scientist, USDA Forest Service, North Central Forest Experiment Station, 208 Foster Hall, Lincoln University Jefferson City, MO 65102-0029. The author wishes to gratefully acknowledge the generous assistance of Darrell Alley, Roderick Davis, Tucker Fredrickson, and Nancy Mikkelson.

Initial reports on the value of tree shelters in silviculture in the United States focused on their dramatic effect on height growth in some species (Lantagne 1991, Potter 1988). But more recently interest has also been directed at ways in which tree shelters can improve survival of planted trees (Marquis 1977, Ward and Stephens 1995). They provide protection and modify the microclimate inside the shelter (Tuley 1984). The use of biostimulants and tree shelters either separately or in combination may reduce the amount of shoot dieback during the early years of establishment, thereby increasing the growth and development of both roots and shoots of young hardwoods. The purpose of this study was to evaluate the impact of preplant root treatments and tree shelters on survival and growth of black walnut and northern red oak seedlings having different root collar diameters and numbers of large lateral roots.

## METHODS

Black walnut and northern red oak were planted separately at the same location. The study was established in an old field in Cedar County, Missouri on a low stream terrace. The soils are in the Helper series (Udollic Ochraqualfs, fine-silty, mixed, thermic). Soils are deep, somewhat poorly drained, and silt loam in texture. Slopes range from 0 to 3%. The field had been in the Conservation Reserve Program for one year prior to planting. The site received no site preparation.

Seedlings for the study were from the George O. White State Nursery located at Licking, Missouri. Prior to planting, root collar diameters and the number of large (>1 mm (0.04 in) lateral roots were measured and recorded for the 1-year-old seedlings. Assigned lateral root groups were (1) 0-4, (2) 5-7, (3) 8-11, and (4) 12 or more. Assigned root collar diameter groups were (small) 9-10 mm, (medium) 11-12 mm, and (large) 13 mm or larger. Seedling roots were pruned to 20 cm in length to aid in planting and bundled in groups of five seedlings of the same species for each assigned lateral root group.

Seedlings were planted in a randomized design at a spacing of 1.5 X 1.8 m. Three replicates per species were planted. One replicate consisted of 4 root treatments X 3 root collar diameters X 2 tree shelter treatments X 4 large lateral root groups X 5 seedlings per bundle. The following root treatments were used: (1) seedling roots soaked in a mixture of supersorb and tap water (71g/18.9 L of water), (2) seedling roots soaked in a mixture of supersorb (71g/18.9 L of tap water) and Roots (0.5 L/9.5 L of tap water), (3) seedling roots soaked in Roots solution (0.5 L/9.5 L of tap water), and (4) seedlings soaked in tap water. Seedlings were allowed to soak for 5 minutes. Half of the original dosage of supersorb and Roots were added to soaking solutions when half of the seedlings for a replicate had been soaked and new solutions were prepared after planting each replicate.

Tree shelters, distributed by TREESSENTIAL of St. Paul, MN, (1.2 m tall and ranged from 7.6 to 10.2 cm in diameter) were placed around seedlings designated to receive them and installed according to distributor's guidelines. Glyphosate was sprayed at the recommended rate in 61-cm-wide strips on both sides of the row within 5 days after planting. Height and diameter measurements and weed control have been done annually. Study data were analyzed using analysis of variance and differences between treatments were tested using the Scheff'e test. Survival percentages were transformed with the arc sine procedure before analyzing.

## RESULTS

### Survival

Root collar diameters at planting ranged from 9.0 to 14.0 mm for black walnut and northern red oak. The survival of black walnut was significantly (p=0.05) affected by the number of large lateral root (Table 1). Survival ranged from 53% for seedlings with 0 to 5 large lateral roots up to 96% for seedlings with 12 or more large laterals. Survival of northern red oak seedlings also increased as the number of large laterals increased, but the affect was not significant. There were no significant interactions between treatments for survival for either species. Both rabbit and deer browsing were apparent for seedlings without shelters. However, the damage did not significantly affect survival after 4 years.

Table 1. Mean survival of black walnut and northern red oak in response to number of large lateral roots[1]

| Large lateral roots Number | Black walnut % | Northern red oak % |
|---|---|---|
| 0-4 | 53a | 49a |
| 5-7 | 75a | 63a |
| 8-11 | 81ab | 60a |
| 12 + | 96b | 68a |

[1]Values in a column followed by the same letter are not significantly different at the 0.05 level according to Scheff'e Test.

Growth

Four years after outplanting, black walnut averaged 75.0 cm in height and 11.3 mm in diameter while northern red oak averaged 39.6 cm in height and 11.2 mm in diameter. The height of both black walnut and northern red oak seedlings was significantly affected by tree shelters (Table 2). Black walnut and northern red oak seedlings in shelters were 36 and 37% taller, respectively, than seedlings without shelters. The interactions of tree shelters with other treatments were not significant (Table 3).

Table 2. Height of black walnut and northern red oak with and without tree shelters[1]

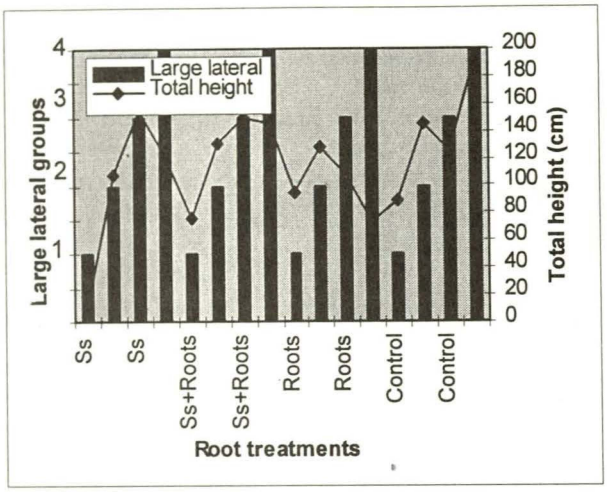
| Tree shelter | Black Walnut cm | Northern red oak cm |
|---|---|---|
| Without | 104.8a | 61.3a |
| With | 163.9b | 97.5b |

[1]Values in a column followed by the same letter are not significantly different at the 0.05 level according to Scheff'e Test.

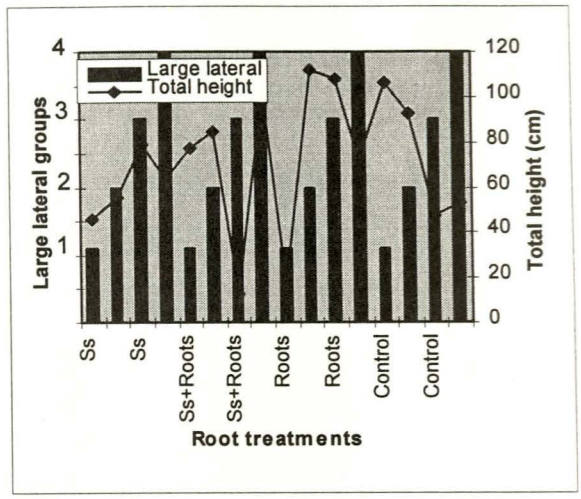
Table 3. Treatments and their interactions with degrees of freedom, mean squares, and p-values for total height of black walnut and northern red oak seedlings 4 years after outplanting

| Source | Degrees of freedom | Mean squares | P-values |
|---|---|---|---|
| | Black Walnut | | |
| Tree shelter (T) | 1 | 147,392 | 0.0001 |
| Root (R) | 3 | 7,146 | 0.0113 |
| TXR | 3 | 128 | 0.9762 |
| Large lateral (L) | 3 | 22,179 | 0.0001 |
| TXL | 3 | 2,816 | 0.2132 |
| RXL | 9 | 9,875 | 0.0001 |
| TXRXL | 9 | 2,307 | 0.2748 |
| Caliper (C) | 3 | 5,072 | 0.7466 |
| TXC | 3 | 540 | 0.8315 |
| RXC | 8 | 578 | 0.9604 |
| TXRXC | 4 | 2,631 | 0.2317 |
| LXC | 4 | 1,301 | 0.5923 |
| TXLXC | 2 | 2,381 | 0.2807 |
| RXLXC | 5 | 725 | 0.8540 |
| | Northern red oak) | | |
| Tree shelter (T) | 1 | 26,435 | 0.0003 |
| Root (R) | 3 | 2,818 | 0.1737 |
| TXR | 3 | 862 | 0.6555 |
| Large lateral (L) | 3 | 4,485 | 0.0562 |
| TXL | 3 | 1,301 | 0.4924 |
| RXL | 9 | 5,331 | 0.0065 |
| TXRXL | 9 | 1,878 | 0.3417 |
| Caliper (C) | 3 | 4,215 | 0.0672 |
| TXC | 2 | 364 | 0.7956 |
| RXC | 6 | 1,504 | 0.4754 |
| TXRXC | 6 | 3,775 | 0.0751 |
| LXC | 6 | 2,150 | 0.2654 |
| TXLXC | 4 | 640 | 0.6708 |
| RXLXC | 6 | 3,718 | 0.0786 |

The effects of root treatment and number of large laterals and their interactions were significant for black walnut height, but only the interaction between these two treatments was significant for the height of northern red oak. For black walnut, except for Roots among root treatments, seedlings with up to 7 (group 1 and 2) large lateral roots, were shorter than seedlings with 8 or more (group 3 and 4) large lateral roots (Figure 1). The shortest walnut seedlings in the Roots treatment were with the highest number of large lateral roots. However, in the case of




BW




NRO

Figure 1. The effect of root treatments and the number of large laterals on the height of black walnut (BW) and northern red oak (NRO) seedlings 4 years after outplanting.

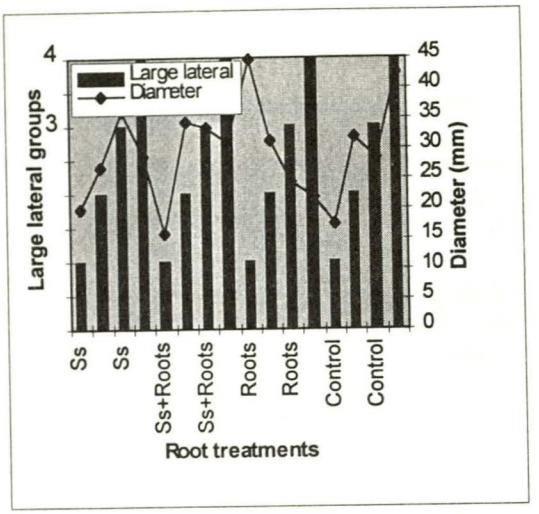
northern red oak, except for seedlings in the supersorb + Roots treatment, the taller seedlings were those with 0 to 7 (group 2 and 3) large laterals. The taller seedlings in the control treatment had 0 to 7 (group 1 and 2) large laterals.

The main effects and their interactions for diameter are shown in Table 4. The use of tree shelters with northern red oak was the only significant main effect for diameter. Northern red oak seedlings with tree shelters had a mean diameter of 10.4 mm compared to 12.4 mm for seedlings without tree shelters. The mean diameter of black walnut seedlings followed a similar trend for the two treatments, but the difference was not significant. Black walnut seedlings with tree shelters had a mean diameter of 27.8 mm compared to 31.8 mm for seedlings without shelters. The interactions for root treatment and number of large lateral roots were significant for both black walnut and northern red oak. Also, the interaction between the number of large lateral roots and root collar diameter was significant.

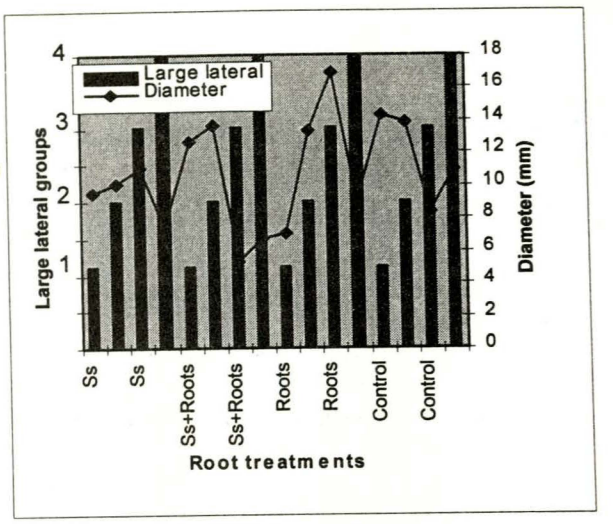
Table 4. Treatments and their interactions with degrees of freedom, mean squares, and p-values for diameter of black walnut and northern red oak seedlings 4 years after outplanting

| Source | Degrees of freedom | Mean squares | P-values |
|---|---|---|---|
| | | Black walnut | |
| Treeshelter (T) | 1 | 687 | 0.1387 |
| Root (R) | 3 | 109 | 0.7868 |
| TXR | 3 | 137 | 0.7234 |
| Large lateral (L) | 3 | 366 | 0.3189 |
| TXL | 3 | 452 | 0.2285 |
| RXL | 9 | 891 | 0.0037 |
| TXRXL | 9 | 296 | 0.4807 |
| Caliper (C) | 3 | 69 | 0.8809 |
| TXC | 3 | 231 | 0.5305 |
| RXC | 8 | 171 | 0.8193 |
| TXRXC | 4 | 277 | 0.4741 |
| LXC | 4 | 422 | 0.2554 |
| TXLXC | 2 | 213 | 0.5072 |
| RXLXC | 5 | 649 | 0.0730 |
| | | Northern red oak | |
| Treeshelter (T) | 1 | 100 | 0.0410 |
| Root (R) | 3 | 53 | 0.0873 |
| TXR | 3 | 34 | 0.2191 |
| Large lateral (L) | 3 | 39 | 0.1762 |
| TXL | 3 | 13 | 0.6219 |
| RXL | 9 | 53 | 0.0313 |
| TXRXL | 9 | 17 | 0.6250 |
| Caliper (C) | 3 | 215 | 0.9001 |
| TXC | 2 | 27 | 0.3029 |
| RXC | 6 | 45 | 0.0892 |
| TXRXC | 6 | 29 | 0.2638 |
| LXC | 6 | 63 | 0.0234 |
| TXLXC | 4 | 4 | 0.9448 |
| RXLXC | 6 | 50 | 0.0789 |

The diameter of black walnut seedlings decreased as the number of large lateral roots increased in the Roots treatment (Figure 2). The diameter of seedlings in the supersorb and supersorb + Roots treatments increased gradually as the number of lateral roots increased with a noticeable decline for seedlings with more than 12 lateral roots. However, for the control treatment, there was an overall increase in mean diameter as the number of large laterals increased. The diameter of northern red oak seedlings increased as the number of large lateral roots increased with a decline at twelve or more for Supersorb and Roots treatments. Northern red oak seedlings in the supersorb + Roots and the control treatments had larger diameters for seedlings with 7 or less large laterals than for seedlings with 8 or more large lateral roots.




BW




NRO

Figure 2. The effect of root treatments and number of large lateral roots on the diameter of black walnut (BW) and northern red oak (NRO) seedlings 4 years after outplanting.

The interaction between the number of large lateral roots and root collar diameter for northern red oak diameter showed that for seedlings with up to 11 large lateral roots, diameter generally increased as root collar diameter increased, but declined for seedlings with 12 or more large laterals (Figure 3).

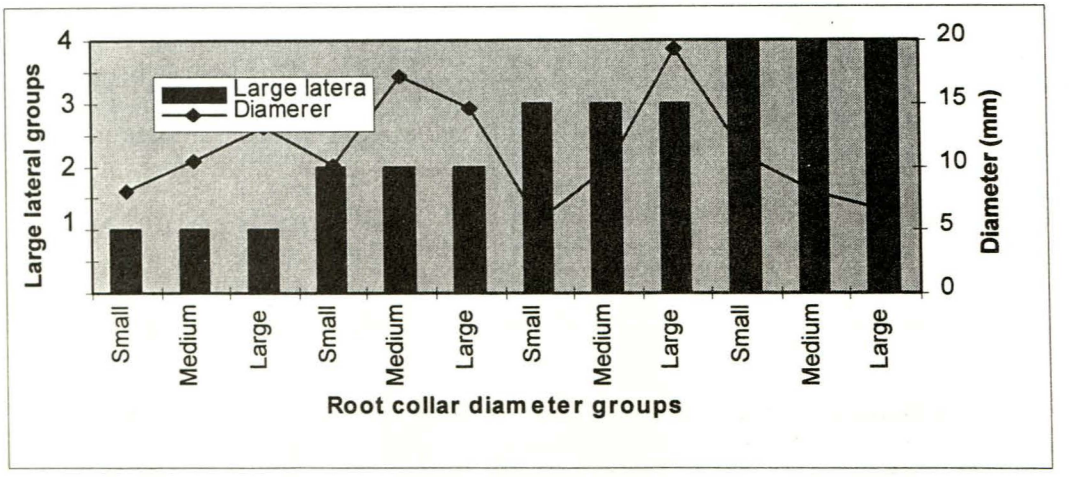



Figure 3. Interaction effect of root collar diameter and number of large lateral root on diameter of northern red oak seedlings 4 years after outplanting

## DISCUSSION

Seedlings with more large lateral roots had better survival (Tables 1). Similar results were reported for seedlings from different mother trees (Ruehle and Kormanik 1986) and for seedlings that were undercut (Schultz and Thompson 1990). The results reported by Ruehle and Kormanik (1986) and Kormanik (1986) also demonstrated that the number of lateral roots on a seedling has a strong genetic relationship to the mother tree. The number of large later

roots can be further increased by undercutting in the nursery and by radicle clipping (Barden and Bowersox 1989, Schultz and Thompson 1990). Undercutting in the nursery reduces shoot growth and improve the shoot:root ratio which makes the seedlings better competitors in their planted environment. Schultz and Thompson (1990) suggested that competitive black walnut seedlings should have large root systems with at least eight or more large laterals and that northern red oak should have at least five large lateral roots.

The significance of the interactions between root treatments and the number of large lateral roots for height and diameter indicates that there is a relationship between them that impact plant growth (Figures 1 and 2). Root absorptive area undoubtedly plays a big role in the response. Supersorb remains active in the soil for only 2 years and the activity of Roots last no more than one growing season. This may not be long enough for growth differences to occur for tree seedlings. Seedlings were excavated to determine if there were root system differences that may be attributed to root treatments.

The difference in height growth often associated with young trees in tree shelters was apparent for black walnut and northern red oak (Table 2). The diameter of seedlings with tree shelters was smaller than for seedlings without tree shelters. Conditions inside tree shelters are characterized by warm temperatures, high humidities (Tuley 1984), lack of air movement, and reduced light intensity; all of which seem to promote height growth.

In conclusion, these data on survival and growth in response to planting seedlings with different numbers of large lateral roots and root collar diameters offer further evidence of how field performance is improved when seedlings with both large root collar diameters and root systems with many permanent large lateral roots are planted. The advantage of using supersorb and Roots as preplanting root treatments for black walnut and northern red oak is not clear in this study. Other studies are in progress to further quantify the benefits of both root treatments and tree shelters on outplanted tree seedlings.

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# IMPACT OF SOIL SCARIFICATION ON THE COMPOSITION OF REGENERATION AND SPECIES DIVERSITY IN AN OAK SHELTERWOOD

James J. Zaczek, Joseph Harding, and James Welfley[1]

Abstract: This study was conducted in a fenced 1-yr-old 70-acre mixed-oak shelterwood to determine the impact of soil scarification on species composition and the production of oak regeneration from abundant northern red oak (*Quercus rubra* L.) acorns. In October 1993, seven replicates were established and randomly divided into control and scarified plots. Pretreatment sampling showed that control and scarified treatments had statistically similar numbers of advance regeneration in various species categories (including total seedlings - 4,085/acre) as well as similar numbers of acorns (48,463/acre). Species composition was dominated by red maple (*Acer rubrum* L.) (45%), followed by oaks (27%), acceptable low value species (24%), and other desirable hardwoods (4%) with 14 tree species found in each treatment. Plots were lightly scarified using a brush rake-equipped crawler dozer in an attempt to incorporate acorns into the soil (scarification seeding) after dissemination but prior to leaf fall.

One year later, the number of seedlings in scarified plots (16,870/acre) was significantly greater than that in control plots (9,614/acre). The number of seedlings increased for all species categories in control plots and for all but red maple in scarified plots. Species composition shifted as a result of treatments with the result that scarified plots totaled significantly more northern red oak (11,596/acre) and fewer red maples (1,106/acre) compared to control plots (1,002/acre, and 3,973/acre, respectively). Composition of other desirable hardwoods (4% of total) and low value species (15% and 11%, respectively) were similar for control and scarified plots. Based on the Shannon-Weiner index, species diversity of the tree seedling community slightly increased from 1.73 to 1.81 in control plots, in part, reflecting more balanced numbers of red maple and oaks. Species diversity in scarified plots decreased from 1.96 to 1.30 reflecting the greatly increased dominance of northern red oak. This decrease occurred in scarified plots despite an increased tree species richness and increased numbers of seedlings in 13 of 15 species suggesting that care must be taken when interpreting diversity values relative to regeneration success. Results show that soil scarification scheduled with recently disseminated acorns prior to leaf fall can significantly improve species composition of seedling regeneration by decreasing red maple and increasing oak seedlings.

## INTRODUCTION

Oaks presently dominate many of the mature upland hardwood forests in the Central Hardwoods Region and in Eastern North America. Oak forests are highly valued for providing essential high-quality wood, wildlife habitat, clean water, varied recreational opportunities, and other resources. In Pennsylvania and in other areas, succession to shade tolerant species, in particular red maple, is occurring in many oak stands in response to disturbance such as gypsy moth (*Lymantria dispar*) induced mortality or cutting (Widmann 1995). The same is true in undisturbed stands except on the most xeric sites (Nowacki and Abrams 1991). The lack of regeneration and failure of oak restoration is a serious problem raising concerns about the composition, quality, and value of future forests.

A key factor in the regeneration of oak hinges on the ability of acorns to successfully germinate and competitively grow with other vegetation. However, relatively few acorns of the total production in a stand ever germinate into seedlings let alone survive and grow to maturity. Steiner (reported in Zaczek 1994) monitoring 2 years of northern red oak seed production in five Pennsylvania mixed oak stands, found that 0.6% of acorns resulted in live seedlings 18 months after dissemination. Acorns are high value food and fall prey to many animals and insects (DeLong 1994, Galford and others 1991, Marquis and others 1976). Environmental factors such as desiccation and freezing

[1] Senior Research Assistant, Forest Supervisor, and Assistant Forest Supervisor, respectively, School of Forest Resources, Pennsylvania State University, University Park, PA 16802.

can seriously impact viability (Olson and Boyce 1971, Young and Young 1992) and surface-seeded acorns are more exposed to lethal cold temperatures and desiccation than those seeded below the soil surface (Bowersox, reported in Zaczek 1994, Sluder and others 1961). DeLong (1994) found that over 2 years less than 1% of planted acorns produced seedlings, and that surface-seeded acorns were located and preyed upon more easily by small mammals.

Scarification using various methods has the potential to incorporate acorns into the soil for added protection. However, results of scarification have been variable depending on the circumstances and methods employed. Scarification has sometimes shown to be beneficial to oak regeneration but impractical to implement on a operational basis (DiMarcello 1986) or to be operationally sound but of little benefit to oak regeneration (Bundy and others 1991). Additionally, little is known about the impact of scarification on species composition and species diversity on treated areas. This study was established to determine if soil scarification would affect the germination of recently disseminated acorns as well as the composition of the regeneration and species diversity in a recently created mixed-oak shelterwood.

## METHODS

### Study Area

This study was conducted in a 1-yr-old 70-acre mixed-oak shelterwood located on the Pennsylvania State University's StoneValley Experimental Forest in Huntingdon County, Pennsylvania. Prior to cutting, the stand was estimated to contain 11,035 board feet of volume per acre and 88.1 $ft^2$ of basal area per acre of all tree species. Of the total board foot volume 76% was oak (ca. 43% *Quercus velutina* Lam., 13% *Q. rubra* L., 10% *Q. coccinea* Munch., 6% *Q. alba* L., and 5% *Q. prinus* L.), 9% white pine (*Pinus strobus* L.), 7% yellow poplar (*Liriodendron tulipifera* L)., 5% Virginia pine (*Pinus virginiana* Mill.), 2% red maple, 1% other hardwoods. The shelterwood was established by whole-tree harvesting the stand in the autumn/winter of 1992/1993 leaving a estimated residual of 50 $ft^2$ of basal area per acre. In the spring of 1993, after harvesting, the shelterwood was enclosed in a 6-wire electric fence.

### Study Design

On 14 October 1993, after acorn dissemination but prior to leaf fall, seven relatively stump-free replicates sized approximately 6 ft x 300 ft were established within a poorly regenerating section of the shelterwood. Because of the virtual lack of natural acorn production within the shelterwood, northern red oak acorns that had been recently collected off-site and float tested for soundness were scattered evenly across the surface of the litter in all of the replicates. Each replicate was divided in two (6 ft x 135 ft) with a buffer zone (30 ft) located between plots. Soil scarified (described below) or unscarified (control) treatments were randomly assigned to the plots in each replicate.

Prior to scarification, 134 circular sample plots (4 ft diameter or 1/3,466.4 acre) were taken at even spacing in each treatment (ca. 19 sample plots per treatment/replicate combination) to estimate the number of acorns and stems of tree regeneration present. Each regeneration stem < 2" diameter at breast height was recorded indicating species and height (classes: 0" - 2", 3" - 12", 13" - 48", >48"). At this time, a representative sample of acorns was collected (n=93), stratified in a refrigerator at 30° F over winter, and sowed in a greenhouse the following spring. Germination was 84%. Herbaceous and woody shrub vegetation was not quantified but visual estimates indicated that there was little present.

The soil scarification treatment was accomplished using a brush rake-equipped crawler dozer (John Deere model 350D). The 6-toothed brush rake was 6.0 ft wide with teeth 1.2 ft apart and 1.0 ft long. The tracks on the dozer were 1.2 ft wide and 2.9 ft apart. The dozer was driven forward with the brush rake lightly scraping the soil surface in an attempt to maintain a scarification depth of 4 inches. After scarification, many of the acorns were not visible and apparently were incorporated below the soil surface. Care was taken not to enter the control plots.

Just over 1 year later, on 19 November 1994, replicates were again sampled for the number of stems of regeneration by species and height class. Approximately 70 circular (4 ft diameter) sample plots were recorded for each treatment.

Data Analysis

Comparisons of the number of acorns present prior to treatment were made between treatments (and replicates) using analysis of variance at the P=0.05 level. For pre- and post-treatment data, comparisons of the total number of stems for the individual species of northern red oak and red maple and species groups (all other oaks, other desirable hardwoods, other acceptable trees, and all seedlings) across all height classes and the total number of stems in each height class across all species were made between treatments (and replicates) using analysis of variance at the P=0.05 level. Other desirable hardwoods include: white ash (*Fraxinus americana* L.), black cherry (*Prunus serotina* Ehrh.), yellow poplar (*Liriodendron tulipifera* L.), and hickory (*Carya spp).* Though some of the following species may be classified as undesirable for various reasons, in this stand, because of their relatively low frequency, we considered eastern white pine (*Pinus strobus* L.), Virginia pine (*Pinus virginiana* Mill.), American beech (*Fagus grandifolia* Ehrh.), sweet birch (*Betula lenta* L.), serviceberry (*Amelanchier arboria* Michx. f.), hawthorne (*Crataegus spp.*), flowering dogwood (*Cornus florida* L.), and bigtooth aspen (*Populus grandidentata* Michx.) as acceptable low value species for maintenance of diversity. Summaries and results are estimates of the number of stems of regeneration presented on a per acre basis. Species diversity of the tree seedling community was estimated using the Shannon-Weiner (H') index (Barbour and others 1987).

## RESULTS

In 1993, prior to treatment, the number of acorns on the ground averaged 48,463/acre and did not significantly differ (P>0.05) between control (49,705/acre) and scarified (47,220/acre) treatments or among replicates. Additionally, areas designated for control and scarification did not statistically differ in the total number of regeneration stems (3,863/acre and 4,308/acre, respectively) or in the numbers of northern red oak, red maple, other oaks, desirable hardwoods, or acceptable species (Figure 1). The number of stems for most of the above species tended to be evenly distributed across the study area as significant differences among replicates were found only for the other oaks and total stems groups. Red maple was the dominant species making up 45% of the total number of stems.

One year after treatment, every species or species group in the control treatment had increased numbers of stems compared to the previous year (Figure 1). In the scarified treatment, most species or groups also increased in number except for red maple which decreased by 652 stems/acre.

Despite the decrease of red maple, the total number of stems of regeneration in the scarified areas (16,870/acre) was significantly greater than in the control areas (9,614/acre). Most strikingly, there were significantly more northern red oak seedlings in the scarified treatment than in the control (11,596/acre vs. 1,002/acre, respectively). However, the scarified treatment also had significantly fewer numbers of other oaks and red maples than the control treatment. Numbers of desirable hardwoods and acceptable species remained statistically similar among control and scarified areas after treatment.

Pretreatment sampling determined that 95% of regeneration was less than 12 inches in height and only 1% was taller than 48 inches which was comparable for control and scarified treatments (Figure 2). Within a height class, both treatments had similar numbers of seedlings except for the 3" to 12" class in which there were significantly more stems in scarified plots (3,343/acre) than in control plots (2,610/acre). A significant replicate effect was also present only in this height class indicating unequal distribution of 3" to 12" stems across the study site. In 1994, after treatment, significantly more regeneration stems in this height class were present in scarified plots (13,354/acre) compared to control (6,320/acre). This occurred without a significant replicate effect indicating relatively even distribution of the number of 3" to 12" stems across the site after scarification. Though control plots had somewhat more regeneration stems in the tallest height class (>48"), 79% of the stems were either red maple or acceptable low value species.

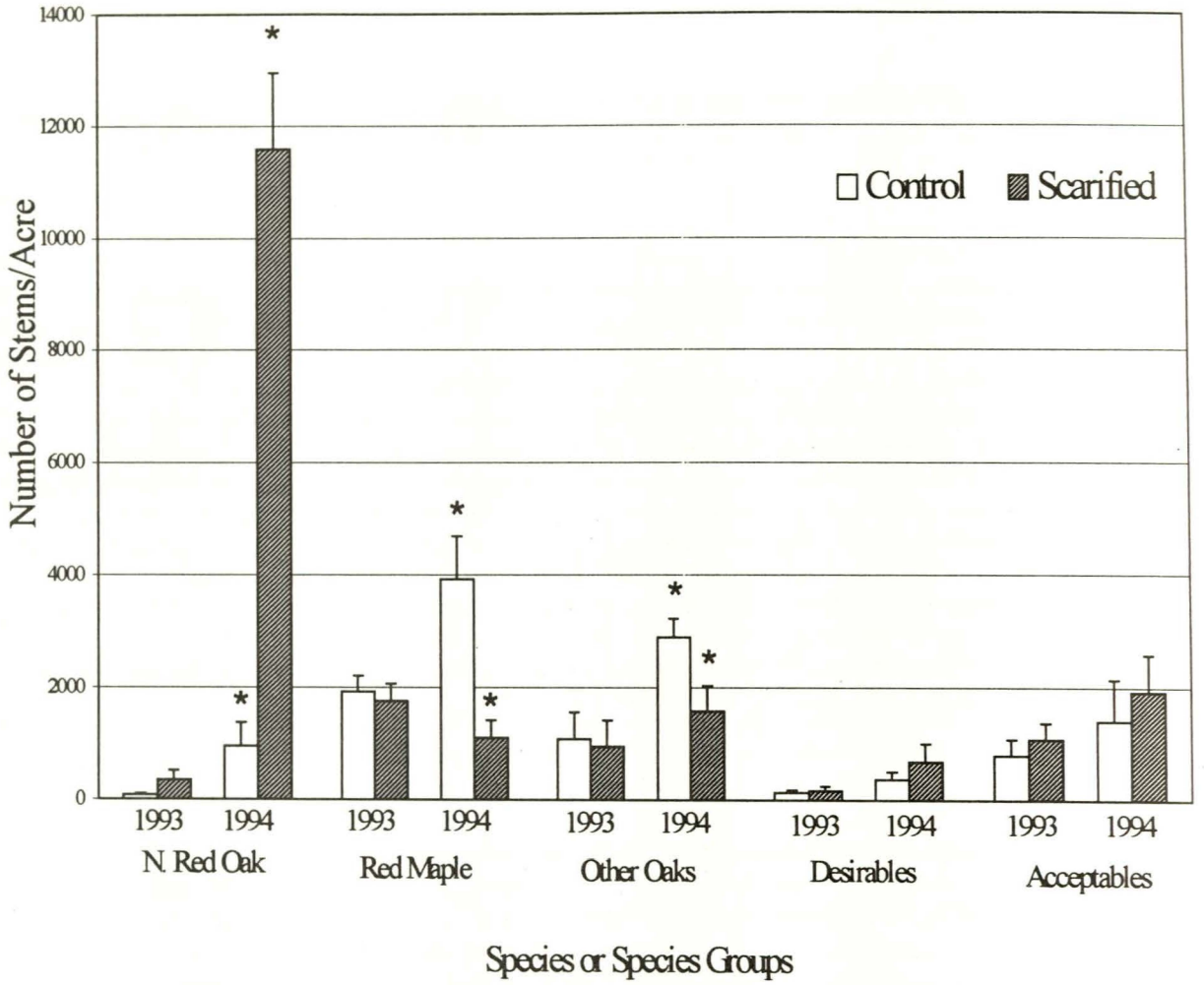


Figure 1. The mean number of regeneration stems per acre (and standard error bars) for species and species groups in control and scarified plots prior to (1993) and after (1994) treatment. Paired bars for a species or species group in a year with asterisks are significantly different at P=0.05.

Species diversity (H') for tree species regeneration prior to treatment was 1.73 and 1.96 in control and scarified treatments, respectively. Species richness was equivalent in both treatments with 14 species present. Following treatment, species diversity in control plots increased slightly to 1.81 while richness decreased to 13 species. In the scarified treatment, species diversity decreased to 1.30 while richness increased to 15 species.

## DISCUSSION

The number of acorns per acre that were disseminated across plots in this study was found to be slightly higher than a 4-year average (41,779) and well within the range (from 540 to 198,510) found previously in five mature uncut mixed-oak stands in Pennsylvania (Steiner 1995). Considering this stand was a shelterwood with a residual of ca. 50 $ft^2$ of basal area of mostly oak, the number of acorns on the ground prior to treatment could be considered somewhat above average but well below the potential maximum oak mast production.

The pretreatment total number of regeneration stems was relatively low (4,085/acre) and generally small-sized. Of the total regeneration, species composition was similar across areas designated for control and scarification and was dominated by red maple (45%), followed by oaks (27%), acceptable species (24%) and desirable hardwoods (4%).

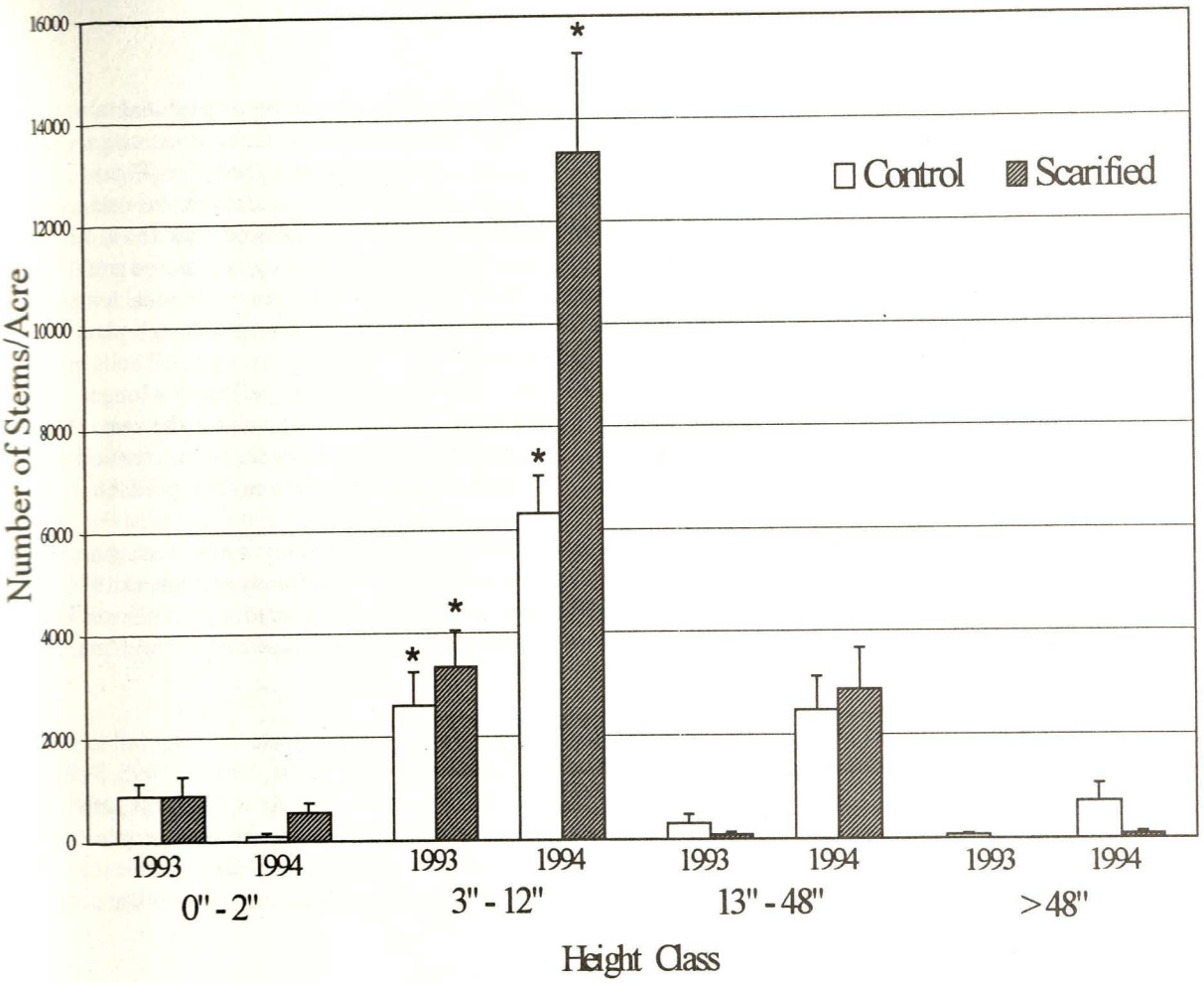


Figure 2. The mean number of regeneration stems per acre (and standard error bars) by height class for control and scarified treatments prior to (1993) and after (1994) treatment. Paired bars in a height class and year with asterisks indicate significant differences between scarified and control treatments at P=0.05.

Even with above average numbers of acorns and the relative lack of red maple seed source within the shelterwood prior to treatment, the regeneration in control plots continued to be dominated by red maple (41%) which more than doubled in number 1 year later. Red maple dominance in control plots may reflect the ability of its seeds to either germinate immediately after dissemination or remain dormant and viable in the soil for at least 1 year until chilling requirements are met (Young and Young 1992). Additionally the wind disseminated nature of the light maple samaras may allow for seed to enter the shelterwood from considerable distances. The numbers of northern red oak stems also increased in control plots but represented the germination of only 2% of the potential viable acorns present at the time of treatment.

One year after treatment, the species composition of regeneration in the scarified plots with 69% oak and 7% red maple more closely resembled the species composition of the trees in the uncut mature stand prior to the shelterwood. In fact, scarification actually reduced the numbers of red maple by 72% while increasing numbers of oaks 12-fold, compared to control plots. The considerable reduction of red maple stems in scarified treatments arises both from a reduction of existing seedlings present before the treatment and from a relative lack of recruitment of new seedlings from seed. This suggests that advance regeneration of red maple may be relatively susceptible to disturbance-induced damage or mortality and that seedbed conditions created by scarification may be less favorable to germination and emergence of its seed compared to undisturbed conditions.

Allowing for the number of pretreatment northern red oak seedlings and the estimated maximum potential rate of acorn germination under controlled conditions, a considerably greater percentage of acorns in the scarification treatment (28%) produced a seedling than was found in control (2%). In a recently harvested red pine (Pinus resinosa Ait.) plantation, DiMarcello (1986) found higher percentage germination of seeded northern red oak acorns in soil scarified with a rototiller than in unscarified plots. In other studies, northern red oak acorns seeded at least 1" below the soil surface showed better germination, were found to be preyed-upon less often by insects and small mammals (Auchmoody and others 1994, DeLong 1994) and experienced less severe micro-environmental temperature and desiccation extremes (Bowersox, reported in Zaczek 1994) than surface-seeded acorns although planting depths below 4" can reduce oak seedling emergence (Smiles and Dawson 1995). Additionally, scarified soils under an uncut hardwood-hemlock stand were found to be between 60° and 100° F warmer and experienced a longer duration of optimum temperatures during spring and early summer compared to unscarified areas in the same stand (Godman and Mattson 1980). Apparently, germination rates can be higher for acorns incorporated into the soil due to improved environmental conditions for overwintering and germination and added protection from predation.

Scarification tended to break stems and uproot some of the advance seedlings resulting either in resprouting or in mortality as has been previously reported using an aboveground scarification technique (Bundy and others 1991). In our study, except for northern red oak, other oaks had reduced numbers of stems in response to soil scarification suggesting that viable seed must be present to germinate into new seedlings to offset the losses that occur from breakage and uprooting of existing seedlings.

Many oak species are episodic seed producers and produce large mast crops every 2 to 6 years and may fail to produce seed in the intervening years (Downs and McQuilkin 1944, Koenig and others 1991, Steiner, 1995, Sork and others 1993). Under natural conditions, acorns are generally considered to be viable only for the first growing season after seed fall (Crow 1988) though a small percentage of planted acorns (1.4%) were observed germinating more than 1 year after sowing (Steiner and others 1990). Thus scarification as a successful seeding treatment to increase numbers of oak seedling regeneration must be thoughtfully implemented to coincide with abundant acorn crops of high viability.

In central Pennsylvania, acorn seed fall is typically completed by the end of October (Zaczek, unpublished data) and large losses to predation can occur rapidly. Steiner (1995) reported that the number of viable acorns remaining on the ground in November was 53.5% of average annual production. Of the 46.5% loss, 38.6% was removed by vertebrates and 7.9% was attributed directly to insect attack or by fungal or bacterial infection after insect damage. Deer predation accounted for nearly half of the removals by vertebrates. This suggests that timing of the scarification treatment should be carried out soon after the majority of the mast crop has fallen to the ground. Scarification after leaf fall could disrupt natural protection by disturbance of the leaf litter layer (Bundy and others 1991). Therefore to maximize acorn germination, optimal timing of soil scarification treatment should be implemented after acorn dissemination but prior to leaf fall.

Planting seedlings or direct-seeding can be used to actively reestablish an oak regeneration component in a harvested mixed-oak stand (Zaczek and others 1993, Zaczek and others in press). Compared to planting seedlings, planting acorns is a more cost effective way of establishing or supplementing oak (Bullard 1992, Countryman and others 1995) on intensively prepared sites or in old-fields. Countryman and others (1995) found that if more than 12% of planted purchased acorns established seedlings, hand planting northern red oak acorns is economically superior to hand planting purchased seedlings. Additionally, they found that machine planted acorns can be from 11 times to 27 times cheaper per acre than hand planted acorns. Since over half the cost of machine planting acorns is for the purchase of seed (Countryman and others 1995), coupling scarification with naturally occurring acorns in a forested situation should result in an even more economical way of establishing oak regeneration.

Based on the decrease in the Shannon-Weiner species diversity values alone, one may conclude that scarification may be detrimental to the genetic diversity and regeneration of the stand. Essentially, the decrease in species diversity for the scarified treatment is a functional result of the tremendous increase in northern red oak seedlings relative to all other tree species. However, scarification increased total numbers of regeneration stems for most species or species groups (including a twelve-fold increase of northern red oak), increased species richness, restored prominence of the oak component and undercut the red maple dominance to levels mirroring the uncut mature

forest. Additionally, scarification also tended to set back the height and reduce numbers of larger well-established advance regeneration of shade tolerant and low value species, perhaps reducing their competitive advantage over more recently established germinants of oak. In this mixed-oak shelterwood with a relatively open canopy, scarification tended to arrest succession and inhibit successional momentum toward shade tolerant regeneration increasing the potential of restoring a significant oak component in the stand. Because of the former results and considerations, we stress that care must be taken when interpreting calculated diversity values relative to the regeneration in managed oak forests.

## CONCLUSIONS

This study demonstrated that soil scarification concomitant with recently disseminated northern red oak acorns significantly increased the oak component in the species composition of regeneration in a mixed-oak shelterwood. Shallow soil scarification is recommended in stands with above average numbers of acorns and low numbers of desirable advance regeneration to effectively plant or scarification-seed acorns to improve germination while effectively disturbing, disrupting, and destroying established stems of less desirable species. Timed application of scarification soon after dissemination but prior to leaf fall should afford maximum protection and improve germination rates of northern red oak mast crops.

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# TWO-YEAR SURVIVAL AND GROWTH OF ARTIFICIAL NORTHERN RED OAK REGENERATION AT GETTYSBURG NATIONAL MILITARY PARK

David S. Larrick, Todd W. Bowersox, Gerald L. Storm, and Walter M. Tzilkowski[1]

Abstract: Overstory competition and foraging by white-tailed deer (*Odocoileus virginianus*) and small mammals are cited as reasons for regeneration failure in mixed-oak stands of Pennsylvania. At Gettysburg National Military Park, deer densities are high (>0.6 deer/ha), and the mixed-oak woodlots were lacking in seedling- and sapling-sized natural oak regeneration. To promote restoration of the woodlots to Civil War condition, we examined the effects of canopy conditions and browsing by deer and small mammals on the survival and growth of planted northern red oak (*Quercus rubra* L.) acorns, seedlings, and saplings. In 1992, three study woodlots were selected as representative of the range of conditions in the Park, including high deer densities. Within each woodlot, three 0.20 ha circular canopy treatments were assigned. Each treatment was replicated three times within each woodlot. Treatments were closed canopy and partial and complete openings in the canopy. Within each canopy treatment unit, two paired fenced and unfenced plots were established, for a total of 18 pairs per woodlot. At each pair of plots, 40 acorns, 8 bare-root seedlings (about 0.3 m in height) and 6 bare-root saplings (> 1.0 m in height) were planted. Survival and height were measured at the end of the second growing season after overstory treatment. About 55% of the 2,160 planted acorns had germinated, and 86% of the 1,196 germinates were alive 2 years after planting. Of the surviving direct-seeded seedlings: 813 were not browsed by deer or clipped by small mammals, 93 were browsed, 105 were clipped, and 13 were browsed and clipped. Woodlot and canopy level were significant predictors of direct-seeded seedling survival, with lowest survival (74%) in closed canopy compared to 86 and 88% for partial and complete openings. Significant predictors of direct-seeded seedling height (mean of 20 cm) were woodlot, canopy level, and fencing. Height significantly increased with increasing amounts of direct sunlight (11, 16, and 30 cm for closed canopy, partial, and complete openings, respectively). Fenced seedlings averaged 25 cm in height, whereas unfenced seedling mean height was 15 cm. Survival for the bare-root seedlings was 93%; there were no significant predictors of survival. Of the 401 surviving stems, 224 were not browsed or clipped, 53 were browsed, 100 were clipped, and 24 were browsed and clipped. Mean height of bare-root seedlings was 35 cm. Significant predictors of seedling height were woodlot, canopy level, and fencing. Height was significantly greater in complete openings (42 cm) than in closed canopy (32 cm) or partial (32 cm) openings. Fencing significantly increased seedling height (31 and 39 cm for unfenced and fenced seedlings, respectively). Survival for the bare-root saplings was 88%. Woodlot was the only significant predictor of survival. Of the 285 surviving saplings, 198 were not browsed or clipped, 77 were browsed, 4 were clipped, and 6 were browsed and clipped. Mean sapling height was 1.44 m. Woodlot was a significant predictor of height. Survival of direct-seeded seedlings increased with increased light levels. Height growth of direct-seeded and bare-root seedlings increased with increased light levels and with fencing.

[1]Graduate Research Assistant, Professor of Silviculture, Adjunct Associate Professor of Wildlife Management, and Associate Professor of Wildlife Science, respectively, School of Forest Resources, The Pennsylvania State University, University Park, PA 16802.

# EFFECT OF TREE SHELTERS ON ABOVE-GROUND STEM BIOMASS

# LEAF NUMBERS AND SIZE, AND HEIGHT GROWTH

Douglas O. Lantagne and Gregory Kowalewski[1]

Abstract: Tree shelters have been tested and shown to be effective in several circumstances regarding hardwood regeneration, especially with northern red oak (*Quercus rubra* L.). A study was initiated to quantify how tree shelters affected quantity, size and biomass of leaves, the number of growth flushes, and the above ground stem biomass of planted northern red oak. Six treatments were established: (1) no tree shelters, (2) standard 4 foot high tree shelters, (3) a 4 foot high tree shelter made of shade cloth (50% shade) and (4) the application of woody brush control on half of each of the three established treatments. Across all treatments in 1995, seedlings averaged 12 inches of height growth during the first flush. Fifty-two percent of these seedlings grew an additional 5 inches during a second growth flush and 16% flushed a third time. In 1996, 24% of the seedlings had four growth flushes. The average growth extensions for each flush were 11, 9, 6, and 3 inches, respectively. Randomly selected seedlings were harvested in 1995 and 1996 to evaluate stem biomass, leaf and bud numbers and leaf area. In 1995, sampled seedlings averaged a height of 50 cm with stems having 17 buds/seedling, 25 leaves/seedling and 38 $cm^2$ surface area/leaf. In 1996, sampled seedlings averaged a height of 79 cm with stems having 60 buds/seedling, 39 leaves/seedling and 42 $cm^2$ surface area/leaf. It appears that plastic tree shelters were beneficial to seedling growth, chemical brush control was marginally effective and shade cloth shelters were ineffective in stimulating growth.

[1] Associate Professor, Michigan State University, Department of Forestry, 126 Natural Resources, East Lansing, MI 48824; and Resident Forester, MSU Kellogg Forest Experiment Station, Augusta, MI, respectively.

# TREE SHELTERS FAIL TO ENHANCE HEIGHT GROWTH OF NORTHERN RED OAK IN THE UPPER PENINSULA OF MICHIGAN

Douglas O. Lantagne and Raymond Miller[1]

Abstract: Tree shelters have been shown to be a questionable establishment practice in shelterwood stands. Experiences with low seedling survival and growth may be due to an apparent deficiency of light. In other situations, tree shelters have generally been found to be beneficial in enhancing survival and growth of hardwood plantings. This poster will describe the poor survival and growth results of a 3 year old tree shelter study with northern red oak in the Upper Peninsula of Michigan. The study was established to test the effectiveness of tree shelters in protecting northern red oak seedlings from deer browse in a heavily used deer area. After 3 years, the northern red oak have not responded favorably to tree shelters. Planted seedlings outside of tree shelters have a higher rate of survival rate and have shown better height growth than sheltered seedlings. It appears that tree shelters on this site were detrimental to the establishment of northern red oak.

[1] Associate Professor, Michigan State University, Department of Forestry, 126 Natural Resources, East Lansing, MI 48824; and Resident Forester, MSU Upper Peninsula Tree Improvement Center, Escanaba, MI, respectively.

# TEMPORAL VARIATION IN WOODY SPECIES COMPOSITION FROM 1922 TO 1996 IN A SECOND-GROWTH APPALACHIAN FOREST

Thomas M. Schuler[1]

Abstract: The *National Forest Management Act*, in part, instructs the Forest Service to maintain the diversity of tree species that are present on federal lands before the onset of management. Further, this act directs the Forest Service to preserve and enhance diversity of tree species within each management area so that diversity is equal to or greater than that of an unmanaged forest. In spite of this directive, it is often difficult to know if management activities are enhancing diversity, maintaining the status quo, or diminishing it. Part of the dilemma is due to the dynamic nature of species composition and the lack of long-term data documenting species abundance with respect to both time and management efforts.

Recently, a forest inventory conducted in 1922 was discovered that permitted an examination of long-term trends in woody species composition. The original inventory was conducted in an area that later became part of the Fernow Experimental Forest in north-central West Virginia. The inventory recorded overstory and understory characteristics that had both second-generation and old-growth components resulting from past harvesting activity (*circa* 1907). Species richness, species diversity, and the actual species present are analyzed with respect to temporal variation. While these mixed mesophytic hardwood stands remain diverse compared with other temperate forests, species richness and species diversity have declined. The virtual loss of American chestnut (*Castanea dentata* (Marsh.) Borkh.) is the most dramatic change in species composition. Less dramatic but significant reductions are also noted for black walnut *(Juglans nigra* L.), butternut (*Juglans cinerea* L.), and American elm (*Ulmus americana* L.). Shifts in species abundance from mid-successional species to both early and late successional species are apparent. A comparison of species diversity related to differing silvicultural methods over the past 50 years in conjunction with management implications is also presented.

[1] Research Forester, Northeastern Forest Experiment Station, USDA Forest Service, Timber and Watershed Laboratory, P.O. Box 404, Parsons, WV 26287.

# ECOLOGICAL CLASSIFICATION SYSTEMS FOR THE WAYNE NATIONAL FOREST, SOUTHEASTERN OHIO[1]

David M. Hix and Jeffrey N. Pearcy[2]

Abstract: The importance of basing land management decisions upon an ecosystem perspective is becoming widely accepted. It is frequently regarded as insufficient to simply manage stands or forest cover types without considering the ecological relationships of the forest vegetation to the other components of the ecosystems, such as soils and physiography. In order to implement the policies of ecosystem management adopted by the USDA Forest Service, ecologically meaningful units of the landscape (ecosystems) must be recognized at various scales or levels. Given this justification, hierarchical ecological classification systems (ECSs) that characterize the range of local ecosystem types, have been developed for the Athens, Ironton, and Marietta Units of the Wayne National Forest. Each ECS was produced separately because of climatic, geologic, and land-use history differences among the three units. The Athens and Ironton Units lie principally within the Western Hocking Plateau Subsection. In contrast, the Marietta Unit is located within the Ohio Valley Lowland Subsection, and was settled first and more heavily farmed. Forest lands of the Athens Unit have experienced the most anthropogenic disturbance, including extensive clay and coal mining. The Ironton Unit was a center of 19th-century iron-ore production, resulting in a majority of the area consisting of even-aged mixed-oak forests.

A multifactor approach was utilized: characteristics of physiography, soils, and vegetation were integrated to identify and classify local forest ecosystems. Intensive field reconnaissance and sampling of a total of 380 sample plots were accomplished over several field seasons, followed by many iterations of multivariate analyses of the different data sets. The majority of the study area within unglaciated southeastern Ohio is comprised of moderately to steeply sloping terrain, however, ridgetops and ravine bottoms were easily distinguished by their level slopes (<16% slope). Canonical correspondence analysis (CCA) indicated that aspect was the environmental variable most highly correlated with different layers of the relatively undisturbed vegetation, for all three units. Therefore, the comparatively mesic northerly aspect ecosystems were separated from those of dry, southerly aspects. CCA was used to confirm which variables were the most important in delineating ecosystem types. Using the national hierarchical framework of ecological units, the most common Ecological Land Types (ELTs) and Ecological Land Type Phases (ELTPs) were defined primarily based upon reasonably stable ecosystem components including landform, slope position, depth to mottled soil, thickness of the A soil horizon, and soil fertility (Ironton only).

Ten major forest ecosystem types have been described for the Marietta Unit, 14 for the Athens Unit, and 8 for the Ironton Unit. The characteristic vegetation of each ecosystem type was specified by distinctive groups of mature canopy trees and of ground-flora species, both identified by two-way indicator species analysis (TWINSPAN). In general, oaks (*Quercus* L) typified the dry and dry-mesic slopes, while other species (e.g., *Acer saccharum* Marsh.) were more common on the mesic and wet-mesic sites. Cluster analysis confirmed the subjective classifications generated in the field, and multiple-response permutation procedures corroborated the differences in vegetation among ecosystem types. These ecosystem classification systems will be useful for managers as they formulate future forest plans, in making land management decisions, and as research frameworks.

[1] Research funding was provided by the Wayne National Forest and the Northeastern Forest Experiment Station, USDA Forest Service; and by the Ohio Agricultural Research and Development Center, The Ohio State University.

[2] Assistant Professor and Research Associate, respectively, School of Natural Resources, The Ohio State University, 2021 Coffey Road, Columbus, OH 43210-1085.

# FIRE HISTORY, POPULATION, AND CALCIUM CYCLING IN THE CURRENT RIVER WATERSHED

Richard P. Guyette and Bruce E. Cutter[1]

Abstract: Quantitative details about past anthropogenic fire regimes and their effects have been lacking in the central hardwood region. Here, we present fire scar chronologies from 23 oak-shortleaf pine (*Quercus* spp. and *Pinus echinata* Mill.) sites in the upper Current River watershed of the Missouri Ozarks. Dendrochronological methods were used to date 2,004 fire scars from 150 shortleaf pine sections with from 100 to more than 300 rings. Fire scar chronologies were constructed from scars dating back 200 to 400 years ago. The average mean fire-free intervals (MFI) calculated for the sites by time period are: the de-populated period, 1580-1700, MFI=17.7 years; the Native American re-population period, 1701-1820, MFI=12.4 years; and the Euro-American settlement period, 1821-1940, MFI=3.7 years. Fire frequency was more variable among and within sites during the Native American period (1701-1820) than during the period of Euro-American settlement (1821-1940). The percentage of sites burned annually before 1850 is correlated ($r=0.87$, $p<0.05$) with population density at the low levels (< 0.64 people per km$^2$). The percentage of sites burned annually decreases about 20% from 1850 to 1940 and is negatively correlated ($r=-0.40$, $p<0.01$) with an increase in population density approaching 4.6 people per km$^2$. About 10 to 15% of the sites burned annually between 1700 and 1800 during a period of low (<0.4 people per km$^2$) population density.

The long-term effects of an anthropogenic fire regime on calcium availability are inferred from Ca concentrations in tree-rings. Calcium concentrations in dated heartwood increments of eastern redcedar (*Juniperus virginiana* L.) for three sites in the watershed are correlated ($r = 0.82$, $p<0.05$) over 340 years with changes in the frequency of anthropogenic fire. Trends in Al concentrations in redcedar heartwood are negatively correlated with the percent of sites burned ($r = -0.70$, $p<0.05$).

## INTRODUCTION

The nature of the past anthropogenic fire regimes of the southeastern Ozarks have long been a question for speculation. Wildland fires influence processes shaping the structure and ecology of these primarily forested natural communities which lie just east of the prairie-forest border (Chandler and others 1983, Paulsell 1957, Pyne 1984, Pyne and others 1996, Wright and Bailey 1982). Long-term records of wildland fires can facilitate our understanding of the present state of the Ozark ecosystems, their wildlife, forest composition, watersheds, and ground flora. Even in the now heavily forested Current River watershed, early travelers described an ecosystem shaped by fire with some grasslands, oak savannas, and open woodlands (Schoolcraft 1821).

In earlier papers, we have described the fire history of a post oak (*Quercus stellata* Wang.) savanna, an oak-hickory (*Carya* spp.) forest, and an eastern redcedar (*Juniperus virginiana* L.) glade (Cutter and Guyette 1994, Guyette and Cutter 1991, Guyette and McGinnes 1982). This study presents results utilizing shortleaf pine (*Pinus echinata* Mill.) remnants to document fire history in the Current River watershed (Figure 1).

The data base for this study was derived from fire scars on shortleaf pine. This species has the largest geographic distribution of any of the southern pines. Over much of this range, despite humid climates, old remnant stumps of shortleaf pine can be found. In the Missouri Ozarks, remnants of shortleaf pine are found in present day oak and oak-pine forests underlain by acid soils (Fletcher and McDermott 1957). While nearly all of the pine and oak-pine

[1] Research Assistant Professor and Associate Professor of Forestry, respectively, School of Natural Resources, 1-30 Agriculture Building, University of Missouri-Columbia, Columbia, MO 65211.

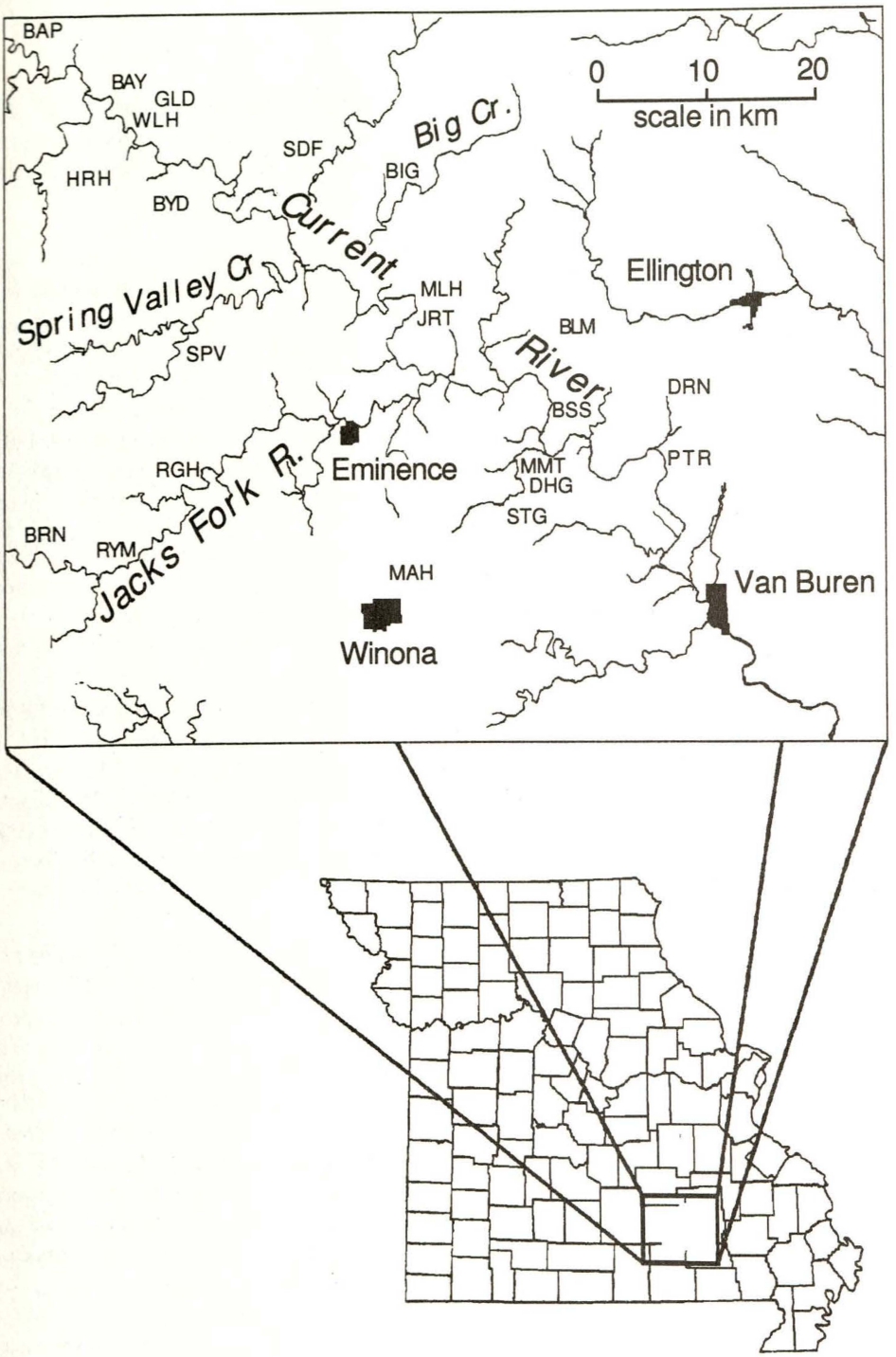


Figure 1. Locations and site codes of fire scar chronologies in the Current River watershed above Van Buren, Missouri. Site codes are identified in Table 1.

forests of this area was cut circa 1900, some old trees, natural remnants, and many old stumps remain. The fire-injured wood of shortleaf pine has been preserved by its high density, hydrophobic oleoresin content, and low moisture content.

In anthropogenic fire regimes of the humid-continental mid-west natural ignitions are rare. It is our hypothesis that gross population levels and culture are important factors influencing the frequency of wildland fire (Dey and Guyette 1996). Hypothetically, fire frequency should increase as the human population and number of potential

ignitions increases. Then, as human population reaches some threshold density, fire frequency should begin to decline as humans have more to from fire. Fire suppression, intensive land use, fuel fragmentation, increased fire breaks, and cultural change all contribute to the decline in wildland fire frequency. The long-term fire histories of the Current River Watershed offer an excellent opportunity to examine these hypotheses.

Wildland fire is also a chemical reaction known to affect the chemistry of soil and plants. There is little evidence, however, of how changes in fire regimes might affect the availability of nutrients over long periods of time. Here we link changes in an anthropogenic fire regime in an Ozark watershed with long-term trends in Ca and Al concentration in dated heartwood increments of eastern redcedar. We discuss possible pathways, reactions, and implications of changes in the Ca budget of low Ca sites.

The four objectives of the research presented here are: (1) to document fire history in a highly dissected forested region of the southeastern Ozarks, (2) develop the use of shortleaf pine remnants for documenting pre-European fire history, (3) test hypotheses that relate human population and culture to fire frequency, and (4) compare the regional fire history with Ca and Al concentrations in eastern redcedar.

## MATERIALS AND METHODS

### Site Selection

The watersheds of the Current and Jacks Fork Rivers are primarily forested (> 80%) and at the eastern edge of a large prairie and savanna mosaic (Figure 1). Shortleaf pine and other species were clearcut from the area between 1880 and 1920 (Cunningham and Hauser 1989). The Current River watershed (above Van Buren), where the sample sites are located, covers about 4316 $km^2$ and is dissected by steep ridges and numerous streams. The surface rocks and outcrops, which were abundant at most sampling sites, were composed of chert, sandstone, rhyolite, and dolomite. The average slope of the sample sites was 18° and ranged from 10° to 32°. Elevations in the study area range from about 140 to 414 m above sea level.

The climate of the study area is humid and continental. Precipitation ranges from 60 to 152 cm and averages 115 cm per year. Spring is the wettest season, followed by fall, summer, and winter. During the fall, winter, and spring of most years, dry warm weather during only a few days may be sufficient to dry ground fuels and permit the spread of ground fires. Severe fires during the growing season are rare but do occur during very hot and dry summers. Natural ignition is rare during most years, despite an abundance of thunderstorms (50-70 thunderstorm days per year) (Baldwin 1973).

Little is know about litter accumulation in oak-pine woods of the Ozarks. Litter accumulation is an important aspect of the fuel matrix in the surface fires of the Ozarks. The decay constant (i.e., the time is takes the litter mass to reach a dynamic equilibrium with site and climate) has important implications for the interaction between fire intervals and fuel accumulation. Scowcroft (1965) quantified litter accumulation in an oak forest of the southeast Ozarks and found that annually burned plots had about a third as much litter as did unburned control plots; plots burned once every 5 years had about 76% of the litter accumulation of the control plots. These rates of accumulation indicate a decay constant of from 6 to more than 12 years. Thus, fire return intervals less than the decay constant tend to reduce fire intensity and spread. Decay rates of litter also effect the among of nutrients, such as calcium, held in the litter layer. The decay constant is an important concept relating fire frequency to intensity, historic fire spread, nutrient pools, and management goals.

Pine stumps and natural remnants were collected from 23 oak-shortleaf pine sites in the upper (above Van Buren, Missouri) Current River watershed. Site names, site codes, the number of trees per site, and the number of scars per site are given in Table 1. Sites were not chosen at random. Sites were located by hiking steep southwest facing slopes scattered throughout the watershed and by consulting with local foresters, wildlife biologists, and land managers. Many areas yielded no remnant pine material. Less than half of the south and west-facing hillsides visited has or had a shortleaf pine component. Collections were made whenever a dendrochronological fire record

(fire scars on pine stumps) was present. Dendrochronological requirements bias the sample to more pyrogenic sites with less fuel moisture for greater periods during the fire season. This may bias the sample sites toward higher fire frequencies on south and west facing slopes where shortleaf pine is most abundant and where stumps are better preserved by a lower moisture content. Thus, these site do not represent MFI for the whole landscape.

Table 1. Site names, map code, calendar years, number trees, and number of dated fire scars for fire and chemistry sites

Fire sites

| Site name and location | Code | Period | Trees | Scars |
|---|---|---|---|---|
| Barn Hollow Natural Area | BRN | 1904-1643 | 5 | 45 |
| Rymers Ranch, NPS | RYM | 1921-1780 | 2 | 26 |
| Manhans, Rocky Cr. S.F. | MAH | 1907-1585 | 9 | 121 |
| Mill Creek | MCR | 1869-1631 | 2 | 63 |
| Paint Rock State Forest | PTR | 1907-1668 | 3 | 64 |
| Stegall Mountain, Peck Ranch | STG | 1941-1636 | 7 | 89 |
| Denning Hollow Glade, | DHG | 1993-1641 | 9 | 102 |
| Mill Mountain, Rocky Cr. S.F. | MMT | 1993-1715 | 11 | 142 |
| Rough Hollow, NPS | RGH | 1921-1558 | 7 | 44 |
| Spring Valley, Alley Sp. S.F. | SPV | 1804-1598 | 1 | 13 |
| Jerktail Mountain, | JRT | 1941-1686 | 15 | 196 |
| Blue Spr., Carr Cr.S.F | BSS | 1993-1626 | 16 | 227 |
| Deer Run S. F. | DRN | 1904-1558 | 13 | 159 |
| Bloom Creek S. F. | BLM | 1914-1773 | 7 | 47 |
| Mill Hollow, Pioneer Forest | MLH | 1993-1656 | 20 | 273 |
| Big Creek, Mark Twain N.F. | BIG | 1950-1608 | 9 | 95 |
| Shannondale S. F. | SDF | 1928-1686 | 5 | 56 |
| Boyds Cr., MO. S. F. | BYD | 1920-1685 | 4 | 34 |
| Hartshorn S. F. | HRH | 1914-1558 | 9 | 101 |
| Welch Spring, NPS | WLH | 1923-1740 | 3 | 21 |
| Gladden Cr., Cedar Grove S.F. | GLD | 1934-1756 | 7 | 40 |
| Bay Hollow, NPS | BAY | 1960-1715 | 2 | 20 |
| Baptist Access, NPS | BAP | 1908-1724 | 2 | 26 |

Stem chemistry sites

| Site name and location | Code | Period | Soil Ca (ug/g) | | | Soil depth | Trees |
|---|---|---|---|---|---|---|---|
| | | | O-Hor | A-Hor | B-Hor | (cm) | |
| Jerktail Mountain | JRT | 1960-1800 | 464 | 114 | 190 | 56 | 1 |
| Williams Mountain | WLM | 1960-1700 | 1112 | 351 | 214 | 91 | 9 |
| Buttin Rock Mountain | BRM | 1960-1620 | 805 | 283 | 135 | 30 | 3 |

Fire Scar Dating

Cross-sections of more than 150 shortleaf pine remnants were cut between ground level and 30 cm. Cross-sections were cut to maximize the number of rings and fire scars. Samples were chosen by the presence of charcoal, fire

scars, and the number of rings present. Compass orientation of the cross-section, slope, and aspect were recorded for each sample. Samples were obtained from cut stumps, wind fallen stems, and standing snags. Wedges were taken from the very few live trees with visible fire scars to extend the fire chronology to the present (Arno and Sneck 1977). Trees ranged in age from 100 to over 300 years. Heartrot and sapwood weathering made determination of the life span of each trees impossible.

Fire scars were identified by callus tissue, traumatic resin canals, and cambial injury. More than 90% of the fire scars were located on the up hill side of the bole. All samples had charcoal present on the scarred exterior. Scars were dated to the first year of cambial injury. Less than 1% of the fire scars occurred during the growing season (period of cambial growth).

Two types of fire scars found on most trees were a single initial scar and subsequent smaller scars. Once the tree has been wounded the thin bark of the callus tissue offers much less thermal insulation to the living tissue. The initial scarring of the bole requires considerable heat to penetrate the bark (Hengst and Dawson 1994, Spalt and Reifsnyder 1962). Lowery (1968), studying cambial injury in shortleaf pine, showed that 15 to 20 min exposure was required to kill the cambium under 4 cm of bark when exposed to air temperatures of 532° C.

The high temperature differences within a fire indicate that initial scarring of trees can be highly variable. Evidence for this variability and the resistance of shortleaf pine to initial scarring comes from old trees and stumps without scars growing adjacent to pine remnants with multiple scars. The bark of one old shortleaf pine, which had no scar, measured 15 cm thick at ground level. Cole and others (1992) found that temperatures in prescribed fires in an oak-prairie mosaic were markedly different within and between ecotypes.

The detailed observations of annual rings and scars necessary for cross-dating require that cross-sections be surfaced to reveal the structure of the rings. Cross-sections were surfaced with a electric hand planer with a sharp carbide blade. Where rings were very narrow or indistinct the ring structure and cellular detail was revealed with sandpaper (220 to 600 grit), fine steel wool, or razor cuts.

A radius (pith to bark ring series) of the cross-section was chosen for measurement with the least amount of ring-width variability due to reaction wood, injury, or callus tissue. The measurement radius was also chosen for the maximum number of rings and high frequency ring-width variance. Often, two radii were measured when ring-width series had much non-climatic variance. Ring-widths series from each sample were measured and plotted. Plots were used for visual cross-dating (Cutter and Guyette 1994, Guyette and Cutter 1991, Stokes and Smiley 1968). Visual matching of ring-width patterns allows the weighing of important "signature years" over years with low common variability among trees. Plots also aid greatly in identifying errors in measurement and missing rings associated with injury or drought.

A computer program, COFECHA (Holmes and others 1986), was used to assist in ensuring the accuracy of both relative and absolute dating of the samples by correlation analysis. Mean between-tree correlation in ring width was 0.51. A floating chronology (undated in absolute time) was established from the samples with the highest common ring-width variance at a site. Absolute dating of the pine remnants was accomplished by cross-dating with a ring-width chronology based on live shortleaf pine growing in Shannon County, Missouri (on record at the International Tree-ring Data Bank, Boulder, Colorado) within the study area and a previously published dated shortleaf pine ring-width series (DeWitt and Ames 1978).

Fire frequency was calculated by site and individual tree locations. Not all trees are scarred in a fire because of the great variation in fuels and bark thickness. Thus, several trees at a site are combined to give a better estimate of fire frequency at a site. On the average it takes about four trees to get a record of most of the fires at a site. This is consistent with Paulsell's (1957) finding that about 25% of the trees at a site were scarred in fires.

Fire intervals for individual trees were also compiled. These fire intervals are not affected by the size of the site considered, or the patchiness of burning in low intensity ground fires within a site. Though these types of data underestimate the actual fire frequency at a site, they are of value as a good estimator of fire frequency and intensity

at point locations. Shortleaf pines are good recorders of fires. This is only true, however, for trees which already have scarfaces. Fires of greater intensity or duration are necessary to cause the initial cambial injury through the thick bark (Cole and others 1992, Gill 1973). This scarring pattern (an initial scar followed by multiple scars) biases the recording of fires by individual trees. To avoid this bias in estimating fire frequency at point locations only the period of record after the first scar was used in calculating fire frequency for individual sites.

### Population Data

Shannon County population data were taken from census data and population estimates in Stevens (1991) and Rafferty (1982) dating back to 1820. Historic Native American population density was estimated from Banks (1978), Stevens (1991), and Gilbert (1996). Shannon County data were used to estimate the change in population density through time in the watershed. Linear interpolation was used to estimate annual population from decadal census data.

Proxy population data were used to relate spatial population differences in the Current River Watershed to differences in fire frequency. These proxy data include the distribution of proto-historic projectile points (Price and Price 1986, Price and others 1983), the distribution of archeological sites (Lynott 1989), and the present day distribution of towns in the Current River watershed.

### Dendrochemical Analysis

Eastern redcedar is an excellent species for dendrochemical analysis of growth increments because the effects of radial translocation of elements within the heartwood are minimal owing to the wood's low moisture content and anatomy (Cutter and Guyette 1993; Guyette and others 1989, 1991). The soils and exposures of the micro-sites on which these trees grow make them particularly sensitive to changes in the availability of mineral elements (Guyette 1994).

Heartwood Al, Ca, soil extractable Ca, and soil pH were measured for 13 eastern redcedar trees and tree sites. These sites are all in the Current River Watershed and overlap, but are not all on fire history sites. Soil sampling and dendrochemical analyses, as well as chronology construction, are described in previous papers (Guyette and others 1989, 1991, 1992a, 1992b; Guyette and McGinnes 1987; Guyette and Cutter 1995).

## RESULTS AND DISCUSSION

### Fire Frequency Among Study Sites

Fire frequency was documented at 23 sites in the Current River Watershed (Table 2). The mean fire-free interval (MFI) was determined for three time periods: 1580-1700, 1701-1820, and 1821-1940. These periods coincide with a de-populated era with few Native Americans, an era of re-population by Native Americans, and Euro-American settlement of the region. In general, the average MFI decreased with time: 1580-1700, 17.7 years; 1701-1820, 12.4 years; and 1821-1940, 3.7 years. Fire frequency was more variable (Table 3) among and within sites during the period of Native American re-population (1701-1820). The range of MFI's is larger for the earlier periods (2.7 to 30 years or more, Table 2) than for the period of Euro-American settlement (1.8 to 8 years). Standard deviations among sites also increase through time. Together this indicates a transition from a spatially variable Native American fire regime to a less variable fire regime after Euro-American settlement. No significant differences were found in the fire frequency of sites on rhyolite versus dolomitic limestone. Effective fire suppression by agencies such as the Missouri Department of Conservation began after 1940 and reduced the percent of sites burned annually from about 21% in 1940 to less than 1% in the 1990's. Chronologies of the percentages of all the trees and sites that were fire scarred in the Current River watershed are illustrated in Figure 2. Some of the years with many scarred sites and trees in the 1600's and 1700's are given.

Table 2. Mean fire free intervals by site and historic period

| Site name | Code | 1821-1940 | | 1701-1820 | | 1581-1700 | |
|---|---|---|---|---|---|---|---|
| | | mean | range | mean | range | mean | range |
| Barn Hollow Natural Area | BRN | 3.8 | (1-9) | 10.0 | (2-36) | . | . |
| Rymers Ranch, NPS | RYM | 5.1 | (1-17) | . | . | . | . |
| Manhans, Rocky Cr. S.F. | MAH | 4.1 | (1-8) | 2.8 | (1-12) | 9.7 | (1-19) |
| Mill Creek | MCR | 3.8 | (1-11) | 3.9 | (1-9) | 7.8 | (1-19) |
| Paint Rock S. F. | PTR | 2.6 | (1-9) | 9.2 | (1-36) | . | . |
| Stegall Mountain, Peck Ranch | STG | 5.0 | (1-10) | 5.0 | (2-12) | 5.9 | (2-11) |
| Denning Hollow Glade | DHG | 2.4 | (1-7) | 7.1 | (1-24) | 30.0 | (6-35) |
| Mill Mountain, Rocky Cr. S.F. | MMT | 2.0 | (1-6) | 8.8 | (1-48) | . | . |
| Rough Hollow, NPS | RGH | 4.2 | (1-8) | 13.3 | (2-57) | 30.0 | (10-73) |
| Spring Valley, Alley Sp. S.F. | SPV | . | . | 13.0 | (8-20) | 20.6 | (11-46 |
| Jerktail Mountain | JRT | 1.9 | (1-4) | 10.9 | (3-21) | . | . |
| Blue Spr., Carr Cr.S.F | BSS | 1.8 | (1-7) | 4.3 | (1-12) | >37 | (.) |
| Deer Run S. F. | DRN | 1.8 | (1-7) | 2.7 | (1-8) | 12.0 | (1-60) |
| Bloom Creek S. F. | BLM | 3.5 | (1-25) | 12.0 | (1-25) | . | . |
| Mill Hollow, Pioneer Forest | MLH | 1.7 | (1-3) | 5.5 | (1-19) | . | . |
| Big Creek | BIG | 2.6 | (1-9) | 6.3 | (1-34) | 18.6 | (8-15) |
| Shannondale S. F. | SDF | 3.5 | (1-13) | 13.3 | (1-21) | . | . |
| Boyds Cr., MO. S. F. | BYD | 4.8 | (1-21) | 20.0 | (2-30) | . | . |
| Hartshorn S. F. | HRH | 2.2 | (1-6) | 5.7 | (1-15) | 7.1 | (1-17) |
| Welch Spring | WLH | 7.4 | (1-33) | 16.2 | (1-55) | . | . |
| Gladden Cr., Cedar Grove S.F. | GLD | 3.7 | (1-20) | >45.0 | (.) | . | . |
| Bay Hollow, NPS | BAY | 8.0 | (1-20) | >50.0 | (.) | . | . |
| Baptist Access, NPS | BAP | 6.3 | (1-15) | 8.8 | (1-47) | . | . |

*Note: Open intervals at the beginning of 3 chronologies with <4 trees during the period are given an interval equal to half of the open-ended time series. Other evidence at these sites, such as live old trees without scars, indicates these sites were not burned often.

S.F. indicates State Forest; N.F., National Forest; NPS, National Park Service

Table 3. Site, fire, and chronology means by historic period and bedrock substrate. Years of record is the sum of the dated rings on all samples. The MFI's of the early periods (1581-1700,1701-1820) are not significantly ($p<0.05$) different from each other but are significantly ($p<0.01$) different than the later period (1821-1940). Different super scripts denote significant differences in MFI's. Rhyolite sites were not significantly different from dolomite sites.

| Period | 1581-1700 | 1701-1820 | 1821-1940 |
|---|---|---|---|
| Number of sites | 11 | 21 | 22 |
| Number of trees | | | |
| (maximum) | 47 | 143 | 148 |
| (minimum) | 4 | 47 | 21 |
| Years of record | 2,254 | 11,167 | 12,616 |
| MFI all sites | 17.7[b] | 12.4[b] | 3.7[a] |
| standard deviation | 10.1 | 11.9 | 2.1 |
| coefficient of variation | 0.57 | 0.96 | 0.57 |
| MFI of rhyolite sites | 18.0 | 7.9 | 2.8 |
| # of sites | 2 | 4 | 4 |
| standard deviation | 17 | 2.5 | 1.4 |
| MFI of dolomite sites | 19.3 | 4.3 | 2.4 |
| # of sites | 4 | 5 | 5 |
| standard deviation | 12.4 | 5.9 | 0.5 |

Scar Intervals on Individual Trees

The distribution of fire-free intervals from single trees (100 to 300 years in age) approximates a skewed normal curve (Figure 3). The most frequent fire intervals at point locations (single-tree site) changed from 4 years between 1580 and 1820 to 2 years between 1821 and 1940. Fifty percent of the fire intervals were below 5.8 years before 1820, while after 1820, 50% were below 4.8 years. This may indicate that fuel-constrained fire intensity limits the scarring frequency of individual trees, since mean fire interval (compiled from many trees) for the associated sites decreased by about 9 years. It is also evident from Figure 2 that the percent of trees scarred annually does not increase as much as the percentage of sites scarred annually. This is an indication that lower amounts of fuels owing to frequent burning are reducing the scarring of individual trees. Fires were more frequent after Euro-American settlement in the 1 to 11 year class and less frequent in the 12 to 50 year class. Figure 3 illustrates the difference (early minus late period) between the percentages of fire intervals on individual trees before and after 1820.

Fire Frequency From Sentinel Pines

A method for estimating the number of fires and changes in fire frequency in a small watershed is the use of "sentinel pines." These are individual shortleaf pines which have a special landscape position relative to fire dynamics. Sentinel pines grow in lose gravel with very little local ground fuel. Despite the lack of ground fuel they have large multiple scars on the uphill side. They grow on the upper slopes of both cherty-dolomite and rhyolite substrate. Long slopes of 15 to 33 degrees with 60 to 240 m of relief typically lie below sentinel pine sites. This landscape position makes these trees good recorders for most of the fires which burn in a small watershed. Pre-heating, turbulence on the lee (uphill side) of the trunk, slope-fire generated winds, and flame length at the top of the

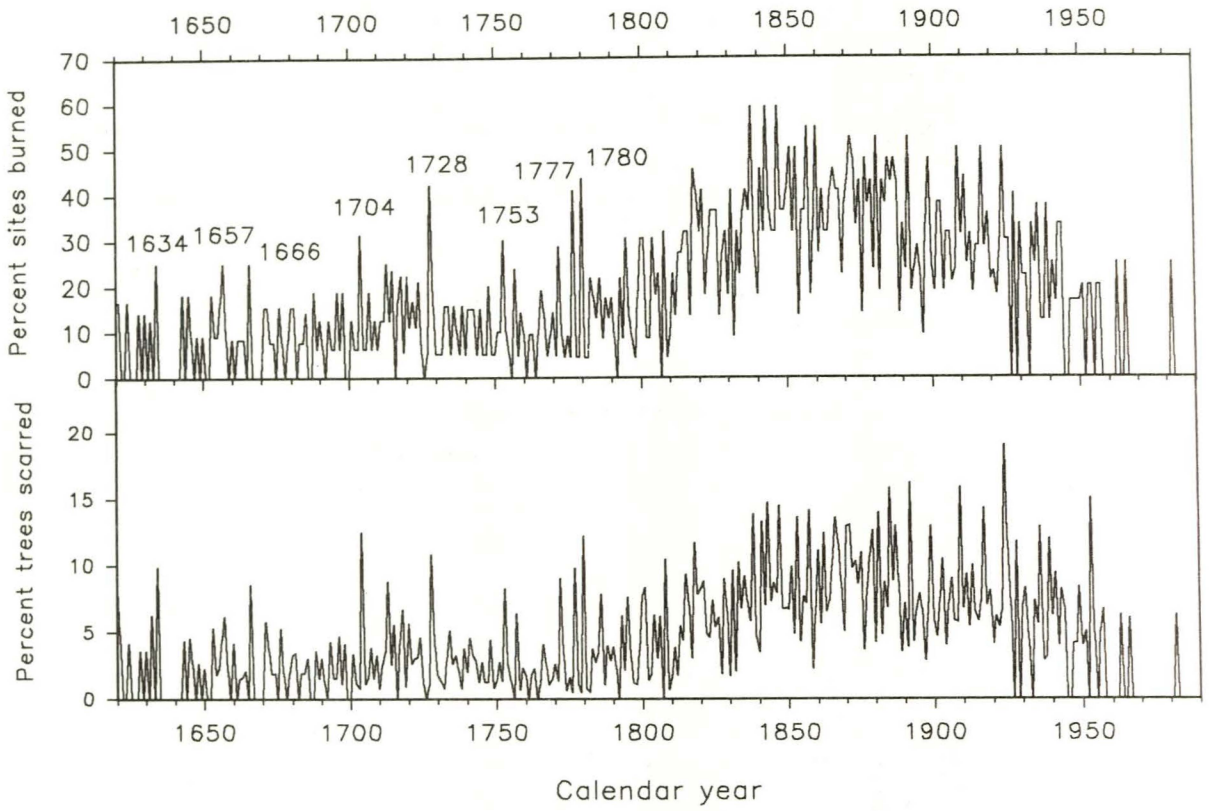


Figure 2. The percentage of trees and sites with fire scars in the Current River watershed. Calendar dates of the years with high percentages of fire scars in the 1600's and 1700's are given.

slope make initial scarring of this thick barked species likely in these locations. These trees are also particularly sensitive to scarring by later fires because of their slow growth and slow healing of wounds. Slow growth, high oleoresin content, high xylem density, and a dry micro-climate, contribute to the preservation of these pine remnants. Because of their landscape position, sentinel pines are excellent recorders of the occurrence of fire in small watersheds. Table 4 gives some basic data on a number of sentinel pines in the Current River Watershed. The mean fire free interval for this group of sentinel pines is 7.0 years for the period (1709-1877). Note that this fire interval, derived from only the 12 trees in Table 4, yields a MFI of 7.0 years that is very close to the average MFI (8.0 years) for the two periods with over 150 trees (1701-1820, 1821-1920; Table 3).

## Population

The fire history of the Current River watershed reveals some widespread tenets about this anthropogenic fire regime. Changes in fire frequency coincide with changes in human populations and culture (Figure 4). Human population is the most important factor affecting fire ignition in much of the central hardwoods region of eastern North America where lightning fires play a minor role in ignitions (Schroeder and Buck 1970).

Old-stock Euro-Americans from the southeastern U.S., mainly Tennessee, began settling in the area circa 1820 (Gerlach 1986). Later, circa 1860, Scotch-Irish immigrated into the area. Fire frequency and the percent of sites burned increased with population density in the Current River Watershed during the 1810-1850 period. Fire frequency is positively correlated ($r = 0.87$, $p < 0.05$) with population at the low population densities (< 0.64 people per $km^2$ ) of this period. The average percent of sites burned in a year increased from 20% in 1810 to 39% in 1850.

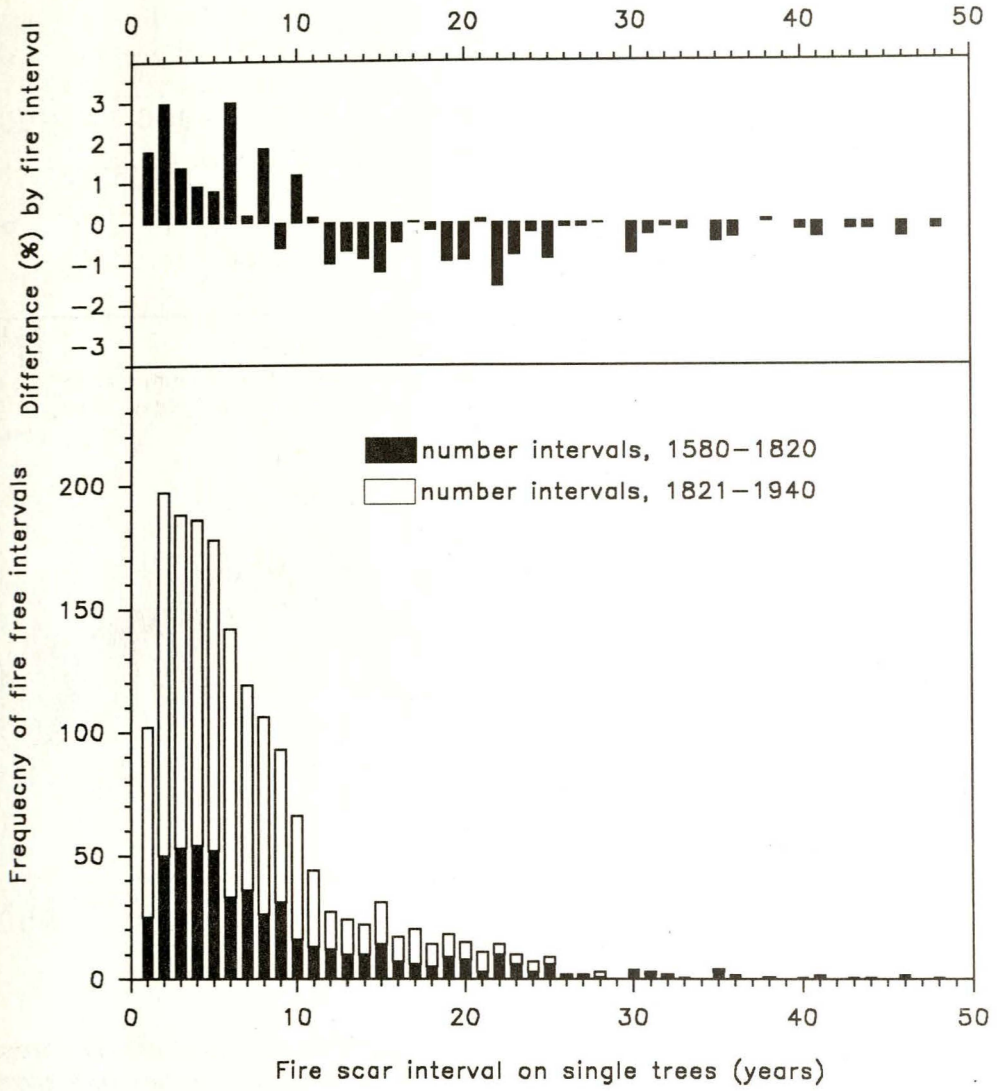
Difference (%) by fire interval
Frequecny of fire free intervals
number intervals, 1580–1820
number intervals, 1821–1940
Fire scar interval on single trees (years)

Figure 3. A frequency distribution (lower graph) of the number of fire free intervals (years between scars) on individual trees. The difference of distributions (upper graph) is the difference between there normalized distributions (1580-1820 %MFI's minus 1821-1940 %MFI's). Note the general shift in the difference of the distributions to shorter intervals (<10 years).

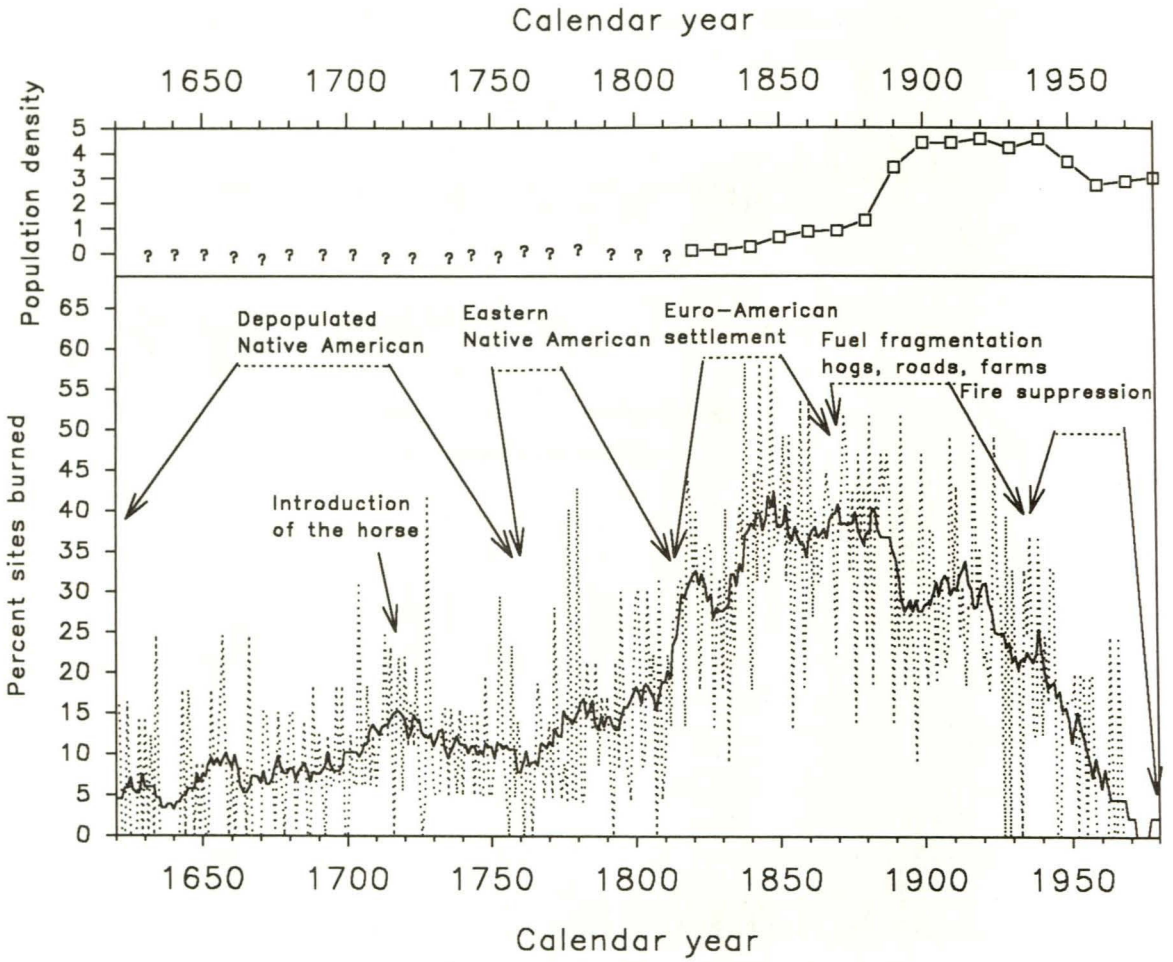


Figure 4. Historic changes in human populations (people per km$^2$) and culture compared to the percentage of sites burned. The dotted line is the sites burned annually (%) while the solid line is an 11 year moving average of the sites burned.

Table 4. Comparison of fire scars on sentinel shortleaf pines from different sites in the Current River Watershed. Sentinel pines are shortleaf pine which occupy sites high on the landscape, with steep slopes, with long distances to the bottom of slopes, and low quantities of local ground fuels. Records are from one tree per site.

| Site | # Scars | Period | Recurrence interval (yrs) |
|---|---|---|---|
| Mill Creek (Middle Current River) | 43 | 1631-1871 | 5.6 |
| Deer Run (Deer Run S. F.) | 20 | 1711-1847 | 6.8 |
| Paint Rock (Paint Rock S. F.) | 25 | 1675-1899 | 9.0 |
| Mahans Creek (Jacks Fork, Rocky Creek S. F.) | 34 | 1748-1870 | 3.6 |
| Stegall Mountain (Peck Ranch, NPS) | 8 | 1645-1695 | 6.3 |
| Mill Hollow (Pioneer Forest) | 26 | 1719-1897 | 6.8 |
| Blue Spring (Carrs Creek S. F.) | 27 | 1747-1887 | 5.2 |
| Mill Mountain (Rocky Creek S. F., NPS) | 25 | 1702-1831 | 5.2 |
| Denning Hollow Glade (Rocky Creek S.F.) | 18 | 1715-1898 | 9.7 |
| Jerktail Mountain (Pioneer Forest) | 29 | 1708-1911 | 7.0 |
| Nordic Hollow (Peck Ranch) | 17 | 1720-1856 | 8.0 |
| Huckleberry Hollow (Cardareva S. F.) | 23 | 1659-1911 | 11 |

At higher population densities (>0.64 people per km$^2$), fire frequency and the percent of sites burned stops increasing with population and begins to decrease. This decrease in the percent of sites burned coincided with increasing population density between 1850 and 1940. A peak in population density (about 4.6 people per km$^2$) was reached in the 1920's. Causal factors for this decline in fire frequency may relate to grazing, fuel trampling by hogs, open range for live stock, roads, some fire suppression, vegetation change, and land clearing.

Population in the Current River watershed has varied greatly over the last thousand years. People of Mississippian Culture populated the rich fertile bottom lands of the Current and Jacks Fork River about 1000 A.D (Lynott 1989, Price and others 1983). These agricultural communities may have supported a relatively large population. Burning for land clearing, defense, game and vegetation management, probably made this a period of high fire frequency. By 1350 these people and their culture left the Current River Watershed.

The end of the Mississippian phase marked the beginning of a long and major de-population of the watershed. De-population caused by disease, warfare, and migration occurred throughout much of North America (DeVivo 1991) and is often linked with changes in fire history (Guyette and Dey 1995, Dey and Guyette 1996). This de-populated era is documented by a decline in the Mississippian substage during which there is only sparse evidence of proto-historic aboriginal people (Price and others 1976). The chronicle of Desoto's exploration in Arkansas mentions that "the only people to the north are those who carry their houses on their backs." During historic times, the Osage traveled from the west to hunt in the Current River watershed (Banks 1978) and the Quapaw lived to the southeast in Arkansas (Chapman 1964, O'Brien 1996).

One remarkable aspect of the early fire record during the de-populated era is that a small human population can maintain a fire regime where about 10 to 15% of the sites are burned annually. Although there are no population estimates of Native Americans in the Current River Watershed before 1820, the 1820's the population density was probably less than 0.10 persons per $km^2$ and much less before the eastern migration of Euro-Americans and Native Americans. In 1820 about 30% of the sites were burning annually. Thus, some very small population (< 0.05 people per $km^2$), whether transient or resident, maintained a fire regime where 10 to 15% of the study sites burned annually. This ratio of population to burning is even more remarkable considering the highly dissected terrain of the Current River Watershed which does not favor for the rapid spread of fire.

Native Americans began migrating back into the Current River watershed in the late 1700's as they were pushed from their eastern lands by Euro-Americans. A general increase in the percentage of sites burned annually in the watershed occurred from 1760 (9%) to 1820 (30%) coincident with the migration of the Cherokee (Gilbert 1996), Delaware, and Shawnee from Eastern North America (Stevens 1991). The Delaware had a history of wildland fire use before arriving in Missouri (Whitney 1994). A 5% increase in the percent of sites burned that occurred circa 1720 is coincident with the acquisition of the horse by the Osage (Waldman 1985).

Artifacts such as willow leaf projectile points, found predominately in the lower reaches of the Current River, indicate some limited use of the area until the late 1700's (Price and others 1986) and a connection to an Armorel cultural phase in northern Arkansas. Although the fire frequency during the earlier part of the de-populated period is generally lower than during later periods it is still much greater than might be expected from lightning ignitions (Schroeder and Buck 1970). Seasonal hunting trips in the Current River watershed by peoples from downstream reaches of the river system may have been an ignition source. The mean fire free intervals for each of the 23 sites during two periods (1701-1820, 1821-1940) is positively correlated (1701-1820, $r = 0.54$; 1821-1940, $r = 0.58$) with the upstream distance of the sites. Less frequent fires in the upper parts of the watershed suggest that these areas were less frequented by humans. Many archeological and historic Native American sites are located in the larger and more fertile bottom lands in the downstream reaches of the Current River (Stevens 1991). Quapaw lived south east of the watershed (Chapman 1964). Also, the Mississippian cultural phase persisted in the southeast Ozark border area until as late as 1700 (Price and others 1976). Even today the lower reaches of the Current River have larger towns than the upper sections. The mean fire free intervals among time periods (1701-1820 and 1821-1940) are significantly correlated (0.54, $p = 0.01$) indicating some continuity in fire frequency through time. Perhaps fertile soils attract more game, hunters, gatherers, and agriculturalist or are correlated with landscape level fire dynamics. All factors that persist to some extent through time. Alternatively, landform gradients within the watershed, such as topographic roughness, may effect the spread and frequency of fire.

Calcium Cycling and Fire Frequency

Parallel trends in the fire frequency of the Current River watershed and heartwood Ca concentrations from three low Ca sites in the watershed were found in the 340 years of common record (Figure 5). The Ca chronology of dated redcedar heartwood increments is highly correlated (Table 5) with the 20 year grouped means of the percent of sites scarred ($r = 0.81$, $p<0.05$) and trees scarred ($r = 0.77$, $p<0.05$). In contrast, aluminum concentrations in heartwood redcedar were negatively correlated (Figure 6) with both the percent of sites scarred ($r = -0.65$, $p<0.05$) and trees scarred ($r = -0.70$, $p<0.05$). Calcium and Al concentrations, however, were not significantly correlated ($r = -0.36$, $p=0.19$) in dated heartwood growth increments.

Table 5. Correlation coefficients among concentrations of elements in eastern redcedar heartwood and estimates of the frequency and extent of fire in the Current River watershed

| Elements | Sites burned (%) | Trees scarred (%) |
|---|---|---|
| Ca (ug/g) | 0.81* | 0.77* |
| Al (ug/g) | -0.65* | -0.70* |
| Ca/Al | 0.75* | 0.79* |

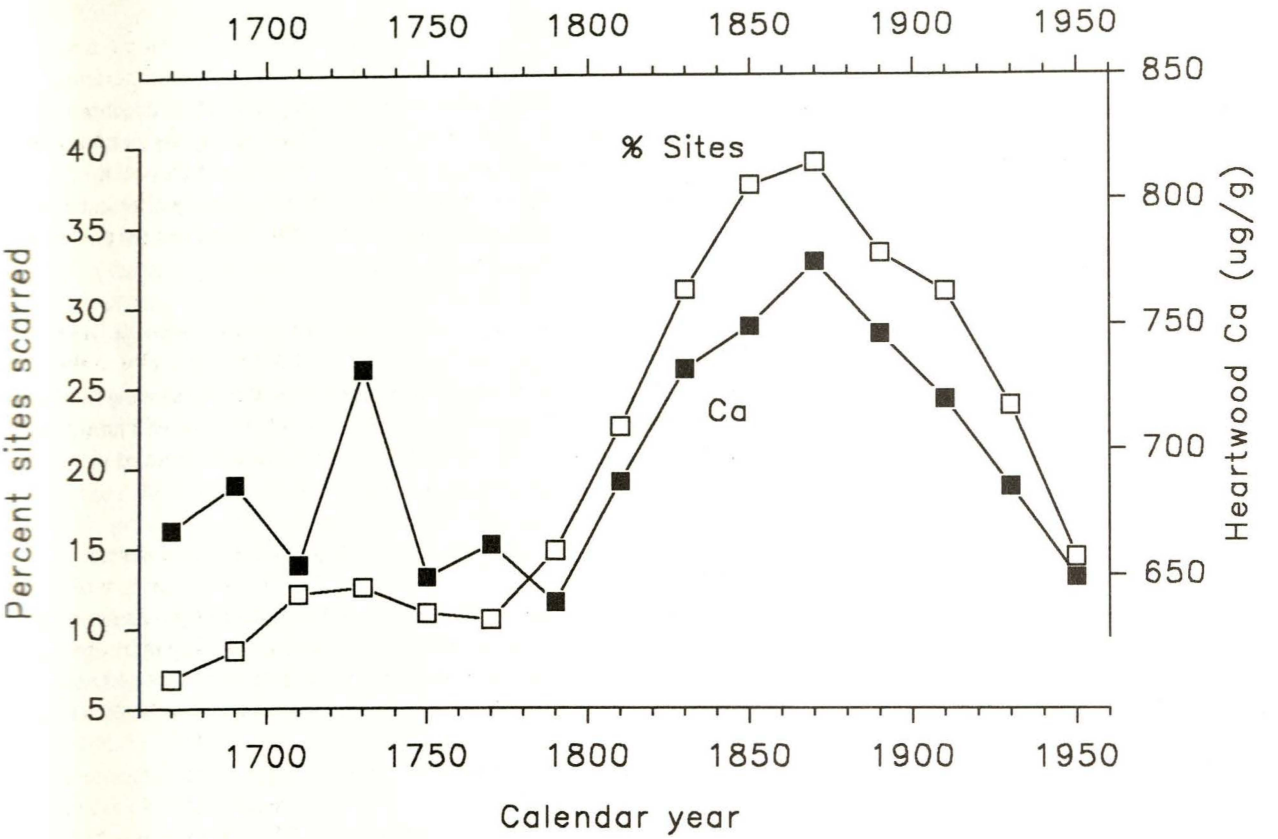


Figure 5. A Ca chronology derived from heartwood increments of 13 redcedar trees compared with the percentage of sites scarred. Annual percentages of sites scarred are grouped and averaged by 20 year periods.

The long-term effects of changes in the frequency of low intensity fires on the Ca availability of soils are not known. The three sites from which Ca chronologies of eastern redcedar were derived have low levels (Table 1) of exchangeable soil Ca (Guyette and others 1992a). Soils from dolomite and limestone sites have from 5 to 8 times the available Ca. These sites, with shallow soils over rhyolite, are an excellent place to examine the long-term effects of changes in fire frequency on Ca cycling because where Ca is scarce its availability is likely to be a factor influencing its concentration in growth increments. Calcium deficiency has been linked to water relations and drought in a number of studies (Lazaroff and Pitman 1966, Tibbitts 1979, Vigouroux and others 1989). Control of Ca uptake in plants via the transpiration stream can result in Ca deficiency. Some redcedar growing at these sites show signs of drought-induced Ca deficiency, such as apical dieback and low sapwood Ca concentrations (Guyette 1994).

Many studies have indicated that Ca is more available to plants after fires. Decomposition of organic matter by fire is both rapid and abiotic. The rate of decomposition by fire is almost instantaneous compared to the rate of decay by biological decomposers. Calcium and other base cations in plant matter are released to ash and leached to mineral soil after fires (Alban 1977) and tend to increase the pH and the concentration of exchangeable Ca (Chandler and others 1983). DeBano and others (1977) found increases in surface soil Ca of 136 kg/ha after a prescribed burn in chaparral. Soil pH is also generally higher in chaparral soils after burning (Vogl and Schorr 1972). Gilliam (1991) found increased Ca availability and an increase in soil pH in the upper 5 cm and 10 cm of soil after a winter burn in an oligotrophic Coastal plain pine flatwoods. Calcium content of the herbaceous layer increased from 2.2 to 5.0 kg/ha. Groesch and co-workers (1991) found increases in soil pH following prescribed burns in Virginia mixed pine forests. Zinke (1991) reviewed the effects of fire on mineral cycling and concluded that Ca and soil pH increase after burns.

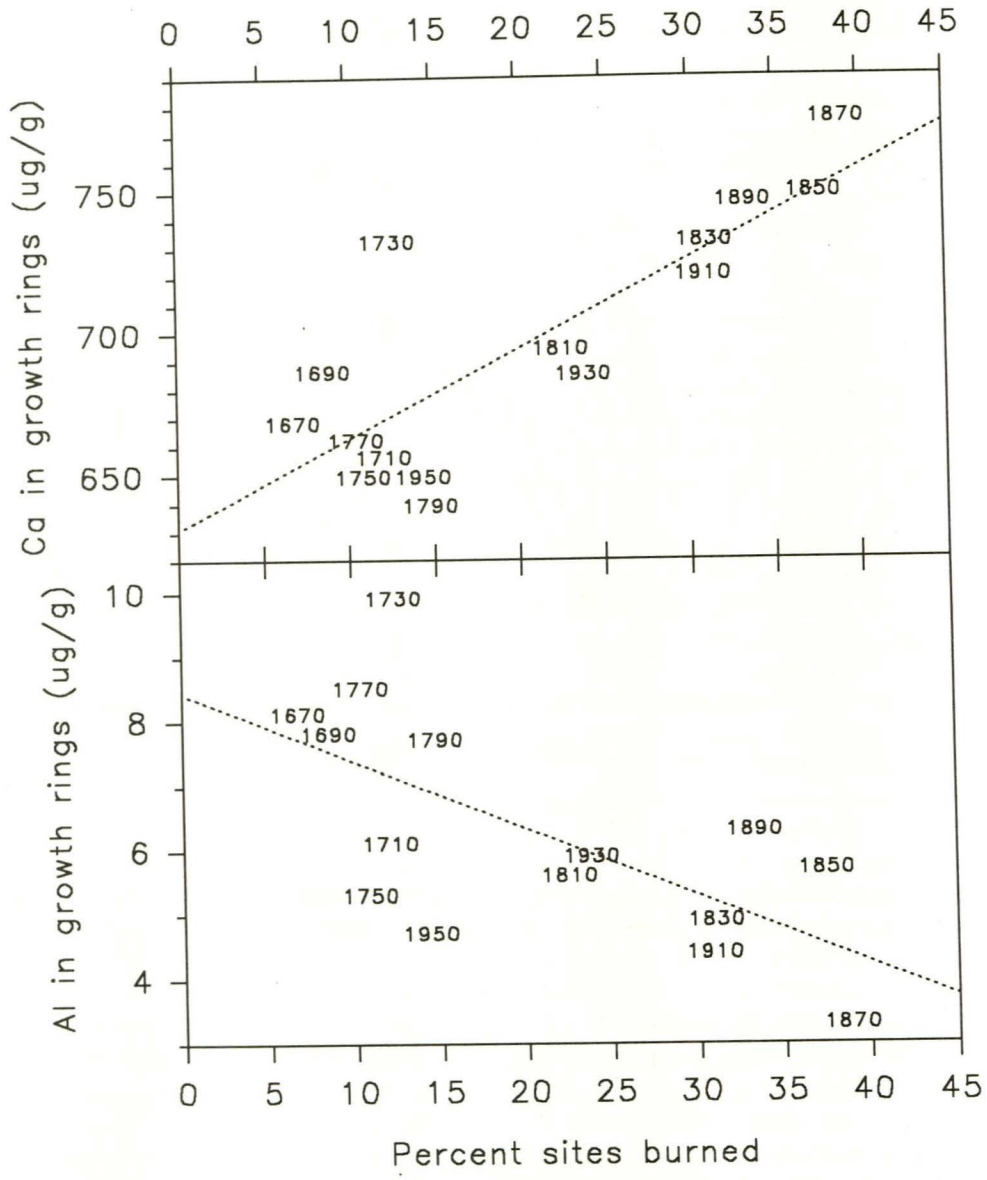


Figure 6. Scatter plots of heartwood element concentrations versus the percentages of sites burned are plotted by date of occurrence. Regression lines are shown.

Guyette and others (1992b) found only an insignificant increase (0.2 units) in a soil pH reconstruction for these sites during the period of high fire frequency (1830-1940). At these sites the acid (mean pH = 4.16) and Al- buffered soils, along with the rapid uptake of scarce soil Ca, may mitigate the effects of the increased rate of Ca cycling on soil pH.

Decreasing Ca cycling owing to reduced fire frequency, coupled with acid precipitation, may limit the availability of Ca in the future on rhyolite knobs in the Ozarks. With more Ca tied up for longer periods in vegetation, plants on these low Ca sites could be more susceptible to drought-induced Ca deficiency in future decades.

## CONCLUSIONS

Fire histories from 23 oak-shortleaf pine sites in the upper Current River watershed of the Missouri Ozarks were reconstructed using dendrochronological methods. Some 2,004 fire scars from 150 shortleaf pine sections were dated to construct fire scar chronologies dating back to 1580. Archeological historical evidence and fire chronologies indicate that the region has gone through three distinct anthropogenic fire regimes during this time. The first, the period from 1580 to 1700, is considered the de-populated period. From 1701 to 1820, Native American tribes settled in the area and used the lands on a continual basis. The final period is from 1821 to 1940 and coincides with Euro-American settlement of the region. Since ca. 1940, effective fire suppression has greatly reduced the scope and intensity of wildfire in this area.

The average mean fire free intervals (MFI) calculated for all sites by time period are: the de-populated period, 1580-1700, MFI=17.7 years; the Native American re-population period, 1701-1820, MFI=12.4 years; and the Euro-American settlement period, 1821-1940, MFI=3.7 years. Fire frequency was more variable within and among sites during the 1701-1820 period. At low population densities ($< 0.64$ people per km$^2$), fire frequency was correlated ($r = 0.87$, $p<0.05$) with the population and settlement of Euro-Americans. At higher population densities (>4.6 people per km$^2$) fire frequency was negatively correlated ($r=-0.40$, $p<0.01$) with population.

The long-term effects of an anthropogenic fire regime on calcium availability are inferred from Ca concentrations in tree-rings. Calcium concentrations in dated heartwood increments of eastern redcedar (*Juniperus virginiana* L.) are correlated ($r = 0.82$, $p<0.05$) over 340 years with changes in the frequency of anthropogenic fire. Trends in Al concentrations in redcedar heartwood are negatively correlated with fire frequency ($r = -0.70$, $p<0.05$)

## ACKNOWLEDGEMENTS

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# DROUGHT EFFECTS ON LEAF ABSCISSION AND LEAF PRODUCTION IN *POPULUS* CLONES

Stephen G. Pallardy[1] and Julie L. Rhoads[1]

Abstract: Leaf abscission and foliation responses to water stress were studied in potted plants of five *Populus* clones grown in a greenhouse. As predawn leaf water potential ($\Psi_l$) fell to -3 MPa, drought-induced leaf abscission increased progressively to 30% for data pooled across clones. As predawn $\Psi_l$ fell below -3 MPa, drought-related abscission was about 50%. When combined with abscission rates in well-watered "phenological control" plants, abscission exceeded 80-90% in the most severely water-stressed plants. Clonal variation in water stress-induced abscission was statistically significant, but appeared to be confounded with clonal differences in leaf loss in control plants. Production of new leaf area ranged from 0 to 1,389 cm$^2$. Pooled across clones, maximum production of leaf area occurred in plants previously subjected to moderate water stress (-0.5 to -0.75 MPa). This stimulation of leaf development was associated with greater areas of individual leaves. At predawn $\Psi_l$ < -2 MPa leaf area production was greatly decreased (<120 cm$^2$). Generally, leaf area production during and after water stress did not vary systematically among clones. However, one clone of *Populus trichocarpa* appeared more capable than others of producing new leaves when severely stressed and allowed to recover. Results indicate substantial sensitivity of *Populus* clones to leaf abscission under water stress, a capacity for refoliation if water stress does not become severe, and some possibility of genetic variation among clones in these responses.

## INTRODUCTION

Poplars are noted for rapid growth and biomass accumulation (Anderson and others 1983, Ranney and others 1987). As such they have received much attention from researchers interested in production of fiber and wood products. Physiologically and morphologically, members of the genus *Populus* are characterized by high transpiration and photosynthesis rates (Bugala 1973, Nelson 1984). They also exhibit prodigious production of leaf area that results from a predominantly free growth habit in which both preformed and neoformed shoots contribute to growth, especially in juvenile plants (Kozlowski and Pallardy 1997).

Many poplars are naturally found on riparian sites or in other situations in which the likelihood of significant seasonal water deficits is relatively small (Burns and Honkala 1990). When exposed to water stress poplars as a group show considerable inhibition of physiological function. Previous research has shown that leaf growth and photosynthesis is greatly reduced even under mild water stress (Braatne and others 1992, Dickmann and others 1992, Heilmann and Stettler 1985, Pezeshki and Hinckley 1982, Regehr and others 1975, Rhodenbaugh and Pallardy 1993). Another drought response of *Populus*, reported anecdotally in the literature, is premature leaf abscission (Dickmann and others 1992, Kozlowski and others 1991, Mazzoleni and Dickmann 1988).

Drought adaptation mechanisms of plants may be classified into broad groups based on responses of plants (Kramer and Boyer 1995). Drought tolerance, the capacity to endure at least modest meteorological droughts, is attained by combinations of physiological responses and morphological and anatomical features that: (1) reduce or postpone dehydration of plant tissues (=dehydration avoidance) or (2) confer the capacity to tolerate dehydration (=dehydration tolerance). Dehydration avoidance adaptations include those associated with acquisition of water resources (e.g., root system extent and patterns of exploration), with efficient internal water transport (e.g., vascular system adaptations), and with adaptations that reduce shoot water loss (e.g., stomatal closure, leaf abscission and high root-shoot ratios). Dehydration tolerance reflects the capacity of a plant to sustain physiological function and/or survive severe tissue water loss. By this classification scheme, poplars generally are considered dehydration-avoiding

[1]Professor and Research Specialist, respectively, School of Natural Resources, University of Missouri-Columbia, Columbia, MO 65211.

species, showing stomatal closure at high leaf water potential ($\Psi_l$) (Regehr and others 1975, Rhodenbaugh and Pallardy 1993), apparent predisposition to leaf abscission under water stress, and little leaf dehydration tolerance capacity (Braatne and others 1992). There are few data available, however, that have specifically related leaf loss to the level of water stress developed in leaves for any species (e.g., Pallardy and Rhoads 1993), and there are no data available for poplars. Additionally, there are few data available that document patterns of leaf production during and after drought in this taxon (Mazzoleni and Dickmann 1988).

Drought tolerance varies genetically within many species and genera of woody plants (see Pallardy 1981 for a review). Some investigations of genetic variation in water-relations-related traits of poplars have been conducted (e.g., Blake and others 1984; Braatne and others 1992; Ceulemans and others 1978; Gebre and Kuhns 1991, 1993; Pallardy and Kozlowski 1981; Rhodenbaugh and Pallardy 1993). However, there have been no comparative studies of leaf abscission responses among different poplar genotypes.

Reviews of productivity in biomass plantations have emphasized the interception of radiation as a primary determinant of dry matter accumulation (Cannell 1985, Kozlowski and others 1991, Landsberg 1986). Water stress may decrease forest stand productivity because light interception by the canopy, which is a function of leaf area index and duration, is reduced. Dehydration avoidance attributes, such as extensive rooting, stomatal closure and leaf abscission are often effective mechanisms of drought tolerance, but they reduce above-ground productivity (Ludlow 1989). On the other hand, dehydration tolerance characteristics would appear to offer the potential for increases in drought tolerance with less inherent reduction in above-ground growth (Kramer and Boyer 1995, Ludlow 1989).

We report in this paper an experiment designed to quantitatively assess leaf abscission and production responses of clones of different genotypes of three species of *Populus*.

## METHODS

The experiment was conducted in a greenhouse at the Forestry Sciences Laboratory at Rhinelander, WI. Eighty 10- to 15-cm long hardwood cuttings of each of two clones of *Populus trichocarpa* Torr. & Gray (T06, T09) and *P. balsamifera* L. (B03, B05) and one clone of *P. deltoides* Bartr. ex Marsh. (D12) were planted in plastic pots holding about 4 L of a potting mix. Slight modification of the basic methods of Parker and Pallardy (1985) and Pallardy and Rhoads (1993) were employed. Briefly, after a period of growth under optimal soil moisture conditions in a greenhouse, plants were allowed to dry slowly by withholding water. As predawn leaf $\Psi$ [as measured with a pressure chamber (Ritchie and Hinckley 1975)] declined, plants were randomly selected and rewatered (Parker and Pallardy 1985). An additional five plants per clone were maintained in a well-watered condition as phenological controls. Leaf abscission was assessed by calculating, at the point rewatering began, the percent abscission of leaves present at the time water was withheld. After rewatering, plants were kept in moist soil for 6 weeks and new leaf area developed since the beginning of stress imposition was measured at this time using a LI-3000 Leaf Area Meter fitted with a belt conveyor (LI-COR, Inc., Lincoln, NE).

Considerable abscission was observed in phenological controls over the course of the experiment, possibly because of the shortening day length during the course of the experiment (August to November) (Addicott 1982). Hence, "drought-related leaf abscission" had to be calculated as the amount of leaf loss that exceeded that of the well-watered phenological controls. Unfortunately, leaf abscission was not assessed in the controls until late September, reducing the number of samples that could be validly analyzed for leaf abscission under drought influence, particularly in mildly-water-stressed plants. There was neither significant leaf abscission nor leaf growth in phenological control plants during the post-drought recovery period for water-stressed plants, allowing full use of samples for new leaf area analysis as a function of water stress.

Data were analyzed by Analysis of Variance (ANOV) using the General Linear Models Procedure of SAS (SAS Institute 1982). Drought-related leaf abscission and new leaf area were analyzed as dependent variables with clone, level of water stress and their interaction as independent variables. Percent abscission was arcsine (arcsin $\sqrt{}$) transformed before analysis to normalize the data (Zar 1984). Additionally, t-tests were conducted on arcsine transformed data to determine if drought-related abscission was significantly different from zero for data pooled

across clones. Leaf abscission of phenological control plants was analyzed using ANOV, but with only clone as an independent variable. Water stress levels were quantified into ranges of predawn $\Psi_1$, with a greater number of stress categories in the low- to moderate-water-stress range in analysis of new leaf area development. The latter categorization was associated with a greater number of samples available in this range for the new leaf area variable.

## RESULTS

### Pooled Data of All Clones

At all levels of water stress imposed, drought-related leaf abscission was significantly greater than zero ($P<0.01$) as determined by t-tests of individual stress level means. Drought-related leaf abscission over all clones increased significantly as predawn $\Psi_1$ declined (Figure 1, Table 1). Leaf loss was progressively greater between 0 and -3 MPa and then increased abruptly below -3 MPa, reflecting nearly total leaf abscission under severe water stress. It is also worth noting that while we did not specifically assess mortality, the proportion of plants showing total leaf kill and complete suppression of new leaf development rose dramatically around -2 MPa (Table 2).

Table 1. Results of Analysis of Variance with models of leaf abscission and new leaf area as dependent variables, and with clone, water stress level and their interaction as independent variables.

| Effect | Drought-related Abscission | New Leaf Area |
|---|---|---|
| Clone | P<0.01 | P<0.54 |
| Stress Level | P<0.01 | P<0.01 |
| Clone x Stress Level | P<0.04 | P<0.42 |

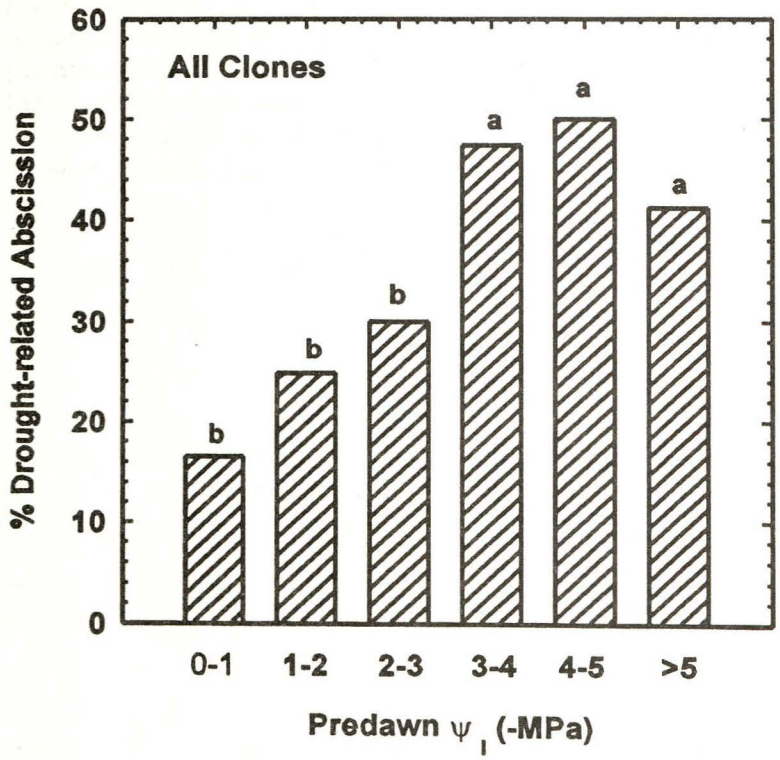


Figure 1. Mean percent drought-related leaf abscission of five *Populus* clones at several levels of predawn $\Psi_1$. Bars not capped by the same letter differ significantly ($P<0.05$).

Table 2. Percent of plants previously dried to several levels of predawn $\Psi_1$ with no living leaves after 6 weeks at high soil moisture as a function of water stress levels. Data are pooled over five *Populus* clones.

| Water Stress Level (MPa) | % Plants Lacking Living Leaves |
|---|---|
| 0 to -1 | 3.2 |
| -1 to -2 | 13.0 |
| -2 to -3 | 56.7 |
| -3 to -4 | 78.6 |
| -4 to -5 | 100.0 |
| <-5 | 87.0 |

Production of new leaf area after the stress period and a 6-week recovery period at high soil moisture peaked at intermediate levels of predawn $\Psi_1$, declining above and below this point (Figure 2). The production of new leaf area under mild water stress was not significantly different from that of the phenological control plants, although it was numerically lower. The production of new leaf area in plants water-stressed to predawn $\Psi_1$ between -0.5 to -0.75 MPa exceeded that in the phenological controls by over 100 $cm^2$; this difference approached the 0.05 level of statistical significance (P=0.09). New leaf area production between plants in the 0 to -0.25 and -0.5 to -0.75 MPa classes was significantly different (P<0.05, Figure 2). Predawn $\Psi_1$ values less that -2 MPa greatly reduced production of new leaf area below that of the control- and mild- to moderately-water stressed plants, and these differences were highly significant (P<0.01, Table 1, Figure 2).

The difference in new leaf area exhibited between mildly stressed and moderately stressed *Populus* plants (Figure 2) was at least partially attributable to the greater number of new leaves produced during the longer drying period necessary to deplete soil moisture in the latter plants (Table 3). However, total new leaf production in both phenological control plants and plants subjected to stress-rewatering cycles was quite similar, differing significantly (P<0.05) only in the most severely water-stressed plants. *Populus* plants water-stressed to -0.5 to -0.75 MPa showed substantially and significantly greater (P<0.05) areas per leaf compared with plants that were more mildly stressed both during and after stress periods (Table 3). Moderately-stressed plants also had significantly greater areas per leaf than phenological control plants.

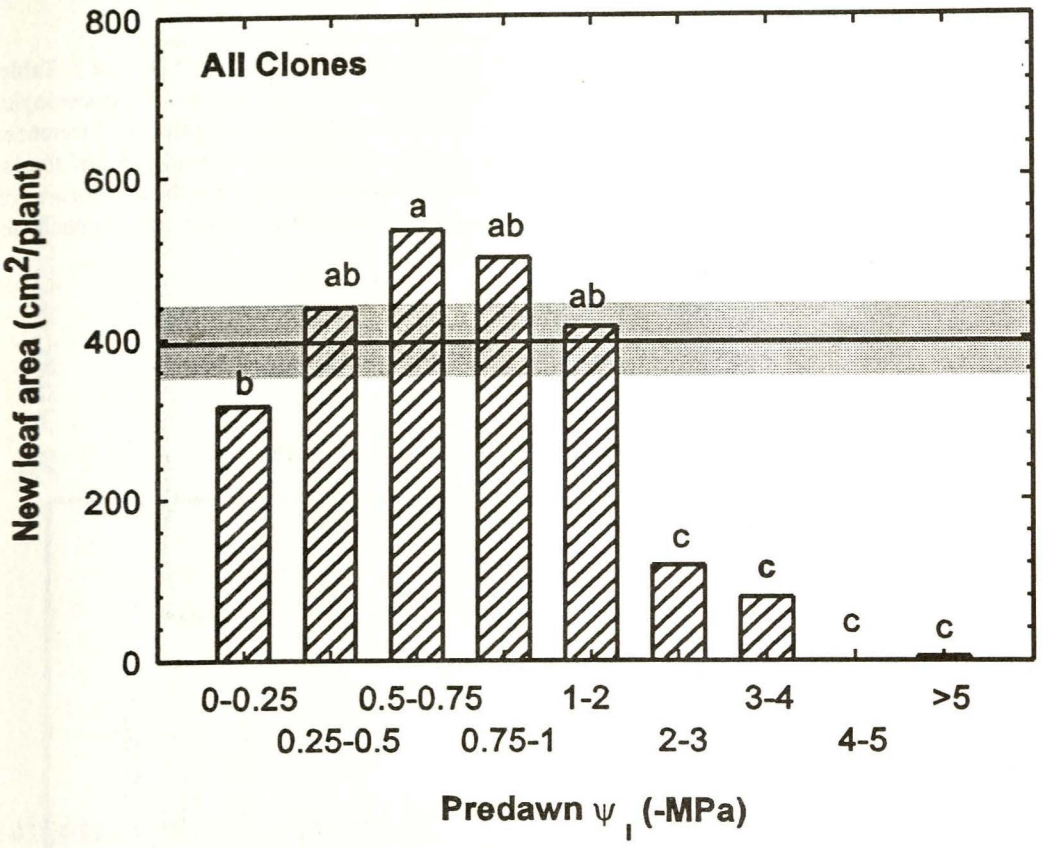


Figure 2. Mean new leaf area production following rewatering of five *Populus* clones previously dried to several levels of predawn $\Psi_I$. Horizontal line indicates mean new leaf area production of phenological control plants pooled across clones. Stippled area indicates ±1 SE of this value.

Table 3. Least squares means of leaf number and area per leaf for plants of *Populus* clones subjected to water stress and subsequently kept well-watered for 6 weeks. No plants fell within the -4 to -5 MPa stress level. Within a columns least square means not followed by the same letter are significantly different (P<0.05).

| Stress Level Imposed | Number of leaves | | | Area per leaf (cm²) | | |
|---|---|---|---|---|---|---|
| | During water stress | After water stress | Total | During water stress | After water stress | Total |
| 0 to -0.25 MPa | 2.63b | 9.38a | 12.00a | 13.78b | 32.90abc* | 32.00ab |
| -0.25 to -0.5 MPa | 2.41b | 8.82a | 11.24a | 15.50b | 35.60bc | 30.31b |
| -0.5 to -0.75 MPa | 5.63a | 6.75ab | 12.38a | 33.44a | 50.32a | 41.94a |
| -0.75 to -1 MPa | 7.22a | 6.89ab | 14.11a | 25.11ab | 39.06ab | 29.62ab |
| -1 to -2 MPa | 5.85a | 7.12ab | 12.96a | 24.55ab | 36.85bc | 32.00ab |
| -2 to -3 MPa | 5.55a | 5.09bc | 10.64a | 18.42b | 23.37bc | 21.30bc |
| -3 to -4 MPa | 6.00a | 6.67abc | 12.67a | 18.06b | 34.43abc | 27.44bc |
| < -5 MPa | 4.67ab | 0.33c | 3.84b | 8.10b | 8.5bc | 6.15c |
| Phenological control | - | - | 12.20a | - | - | 29.54b |

* Difference between 0 to -0.25 MPa and -0.5 to -0.75 MPa levels significant at P=0.06.

Clonal Differences

Statistical differences among clones in drought-related leaf abscission responses were evident (Figure 3, Table 1), but interpretation of these data was complicated by substantial clonal variation in abscission of the phenological control plants (Figure 4). Although analysis of variance of these latter data indicated no significant differences ($P>0.05$) among clones in leaf abscission among the control plants, clones with smaller amounts of leaf abscission (especially clones B03 and T09) tended to exhibit significantly greater levels of drought-related abscission (cf. Figures 3 and 4). Hence, clones B03 and T09 might be considered more susceptible to water stress-induced leaf abscission, but this pattern must be interpreted cautiously.

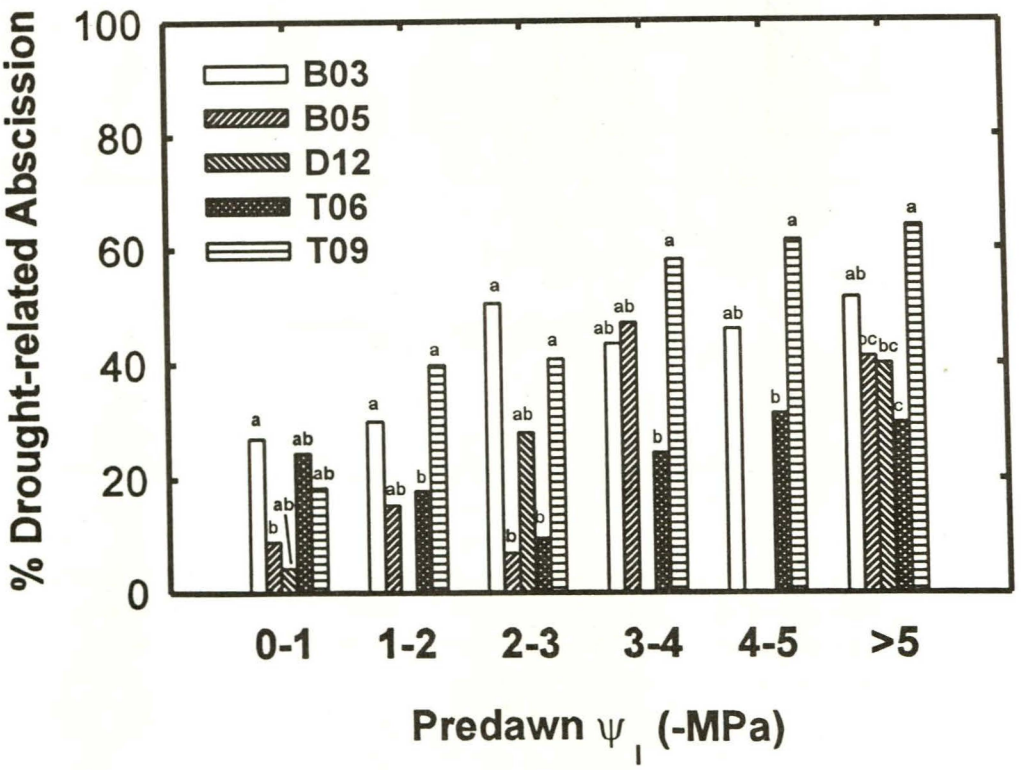


Figure 3. Clonal mean percent drought-related leaf abscission at several levels of predawn $\Psi_1$. Bars not capped by the same letter are significantly different ($P<0.05$).

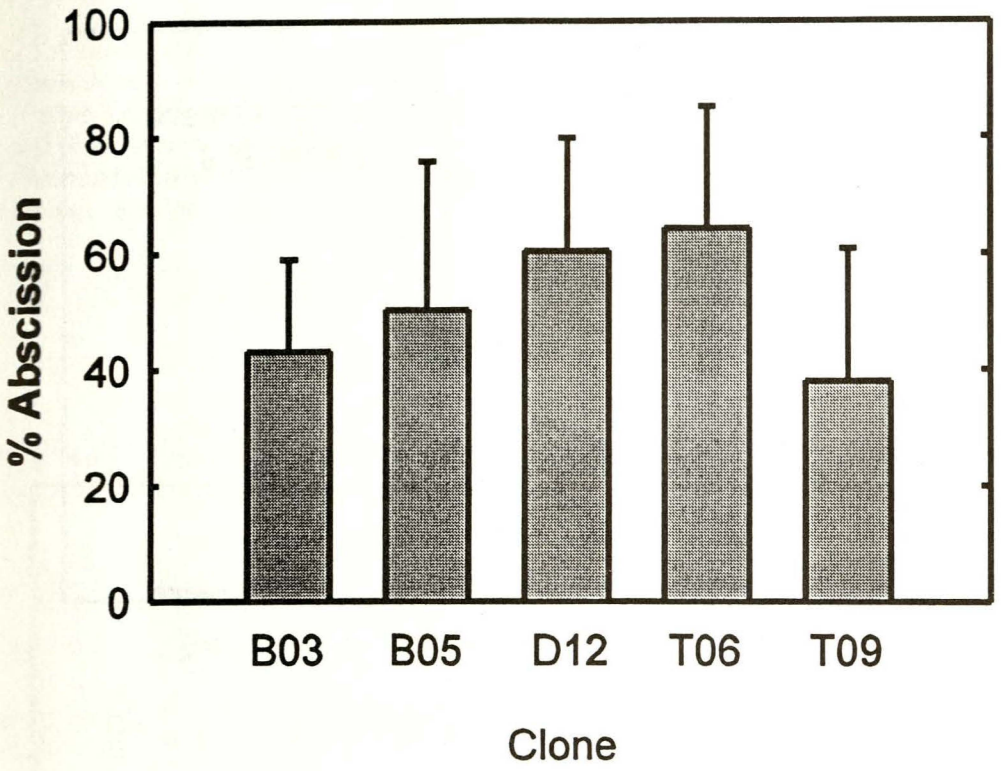


Figure 4. Mean percent abscission of phenological control plants of five *Populus* clones. Bars = 1 SD.

For the most part, new leaf area production during after stress did not differ among clones in any sustained fashion (Table 1). All clones showed capacity to produce substantial new leaf area to predawn $\Psi_1$ values of -2 MPa (which would be characteristic of a moderate to severe drought). When predawn $\Psi_1$ declined further, leaf area production showed a similar decline. There was one possible exception to this trend—Clone T06 appeared to possess the capacity to produce substantial amounts of new leaf area even when stressed severely (Figure 5). However, this pattern was not statistically significant (Table 1). There were no significant differences among clones nor was there a significant water-stress x clone interaction in the number of new leaves produced or in area per leaf (data not shown).

## DISCUSSION

*Populus* clones exhibited enhanced leaf abscission under imposed drought. The levels of predawn $\Psi_1$ shown to cause enhanced abscission in the present study can be expected during seasonal droughts in many areas where unirrigated *Populus* are grown, confirming field observations of the predisposition of this group to abscission (Dickmann and others 1992, Pezeshki and Hinckley 1982). The region of predawn $\Psi_1$ around -2 MPa appears to be a critical threshold for *Populus*, as abscission accelerates with greater stress and production of new leaf area is greatly suppressed.

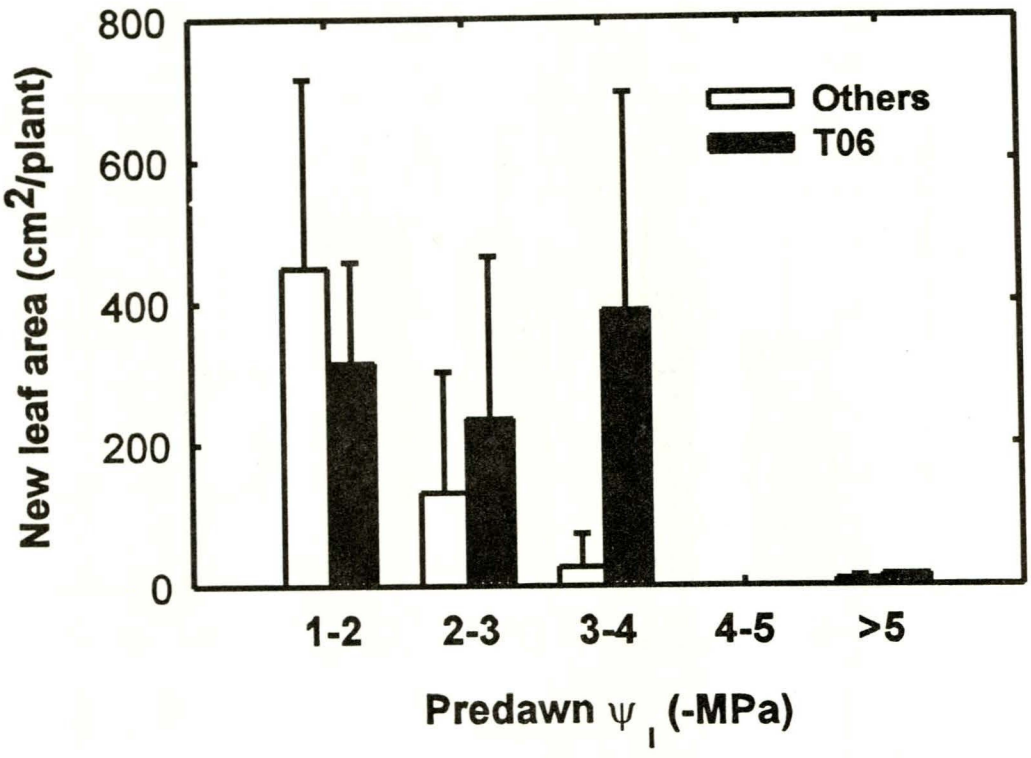


Figure 5. Mean new leaf area production rewatering of plants of clone T06 and all other clones previously dried to several levels of predawn Ψ1. Bars = 1 SD.

Other physiological processes in poplars appear to be correspondingly sensitive to water stress. For example, photosynthesis in two of the clones studied here (B03 and T06) was greatly reduced as $\Psi_l$ of leaves declined from -1 to -2 MPa (Rhodenbaugh and Pallardy 1993). In other more drought tolerant species, such as many *Quercus*, photosynthetic rates and leaf retention are sustained to substantially greater levels of water stress. In post and white oaks (*Quercus stellata* Wangenh., *Q. alba* L.) $\Psi_l$ of -3 MPa or below was required to completely inhibit photosynthesis (Ni and Pallardy 1991, 1992). Even at such low $\Psi_l$ values there was little tendency toward leaf abscission in these two species (Pallardy and Rhoads 1993). In contrast, in the same experiments a frequently riparian species (black walnut, *Juglans nigra* L.) showed patterns of photosynthetic inhibition and leaf abscission that were similar to *Populus* in the present experiment. These responses have obvious implications with regard to competitive ability on dry sites. The general restriction of these light-demanding plants to moist sites may be at least partly attributable to their poor prospects for adequate carbon balance in drought-prone habitats.

As might be expected, new leaf area production was greatly reduced if plants were severely water stressed. Interestingly, moderate water stress appeared to stimulate production of leaf area above that observed in well-watered phenological control plants, perhaps constituting a compensatory response to loss of leaf area via abscission. In the present study, the effect of moderate water stress appeared to be larger leaves, not an increase in number of leaves. Mazzoleni and Dickmann (1988) observed a similar pattern of stimulation of leaf growth in water-stressed *Populus* x *canadensis* 'Eugenei' plants after rewatering. There are obvious limits to this response, as new leaf area development in plants subjected to lower predawn $\Psi_l$ values was greatly suppressed. We also observed a similar pattern of new leaf area development in response to water stress in *Juglans nigra*, but in this species leaf production was extended to -3 MPa (Pallardy and Rhoads 1993). Compensatory leaf area production might be advantageous under a regime of mild-to-moderate drought, as new, physiologically vigorous leaves could increase whole-plant photo-

synthesis. However, it should be remembered that this response attends the loss of existing leaf area that represents a significant loss of carbon and nutrient resources.

Clonal differences in leaf abscission and leaf growth responses were not well defined. Previous studies have shown genetic variation within *Populus* for a wide range of physiological responses (see Introduction). Additionally, genetic differences in autumnal leaf abscission patterns have been demonstrated (Barnes 1969). Although we detected statistical differences in clonal leaf abscission responses to water stress, the differences in abscission responses in control plants confounded meaningful comparisons of water stress responses among clones. More study is needed to confirm the putative differences displayed in the present experiment. There were no statistical differences among clones in production of new leaf area, but it is worth noting that one clone—T06—appeared to exhibit some capacity for sustained leaf production under severe water stress.

## ACKNOWLEDGEMENTS

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# RED MAPLE DEVELOPMENT IN MIXED HARDWOOD STANDS IN WEST VIRGINIA

Brian Tift[1]

Abstract: Recent evidence has shown that red maple (*Acer rubrum*) may become a more prominent species in eastern mixed hardwood stands. This study will examine the strategies used by red maple to reach dominant and codominant canopy positions on two different stands (one mesic and one drier). Stand level data such as dbh, height, crown class, height class, and age structure will be gathered using 0.02 ha plots covering one-tenth of the area of each stand. Six plots per stand, which must contain a codominant red maple, will be randomly selected as stem analysis plots. One codominant red maple and all competing trees will be felled and stem disks taken at ground level, breast height, and every 1 m thereafter. Sub-canopy red maples will also be felled to determine age and periods of suppression. Possible height growth strategies that will be looked for include: (1) advanced regeneration (red maple maintains initial height advantage over competitors), (2) red maple invades at the same time as competitors and grows in height at an equal rate, (3) red maple invades at the same time and reaches the canopy through a series of releases following suppression. The effect of regeneration type on current canopy position will also be examined.

[1] Graduate Research Assistant, Division of Forestry, West Virginia University, P.O. Box 6125, Morgantown, WV 26506-6125.

# COMPENSATORY MECHANISMS OF CENTRAL HARDWOOD FOREST COMMUNITIES IN A CHANGING ENVIRONMENT

Shibu Jose and Andrew R. Gillespie[1]

**Abstract:** Although it is known that forest productivity is highly correlated with canopy leaf area, the influence of environmental gradients on leaf area-productivity relationships has not been well-documented, particularly for mixed-species forests. The present study was conducted to define the mechanisms by which Central Hardwood forest communities respond to changes in environment. Ecological Land Type Phases (ELTPs) of the Central Hardwood region, delineated by soils, topography, and indicator vegetation, provided a framework to examine the impacts of environmental gradients on leaf area-productivity relationships at a landscape level. Three ELTPs— (1) *Quercus prinus/Carex picta-Vaccinium*, Dry slope, (2) *Quercus alba-Acer saccharum - Parthenocissus*, Dry-Mesic slope, and (3) *Fagus-Acer saccharum/Arisaema*, Mesic slope—were selected and stratified into three slope positions viz. the upper edge of the community, the middle or center of the community on the slope, and the lower edge of the community. Circular plots of 0.02 ha in size were established for sampling. Density (standing basal area, volume, and biomass), annual production, relative growth rate, leaf area index, and growth efficiency were determined for each slope position and examined for compensating mechanisms for drier, less fertile conditions. Among the three communities studied, dry slopes compensated for drier conditions through both a shift in species composition and reduced annual production and leaf area. Dry-mesic slopes exhibited adaptation only through changing species composition whereas mesic slopes compensated for drier conditions primarily through reduction in annual production and leaf area. These results indicate reduced productivity for Central Hardwood forests under warmer, drier conditions that have been predicted for the Central U.S. by most general circulation models.

[1] Research Assistant and Professor of Silviculture, respectively, Department of Forestry and Natural Resources, 1159 Forestry Building, Purdue University, West Lafayette, IN 47907

# PATTERNS OF NORTHERN RED OAK GROWTH AND MORTALITY IN WESTERN PENNSYLVANIA

J. R. McClenahen[1], R. J. Hutnik[2], and D. D. Davis[3]

Abstract. This study evaluates the extent and cause(s) of an observed decline of northern red oak (*Quercus rubra* L.) on a 50-km portion of a major anticlinal ridge in west central Pennsylvania, and illustrates an approach for evaluating tree declines. Long term annual observations of forest health revealed the onset of crown dieback in 1983, chiefly among red oak, followed by further dieback and mortality. A plot survey in 1994 within 10 stands where increment cores from healthy red oak were obtained in 1986 revealed higher mortality among red oak than other species, higher mortality on ridgetop sites, greater mortality in the northern than southern half of the sample area, and greater mortality among larger diameter red oak. Radial growth and last growth year determined on cores from dead red oak in the 1994 sampling showed highest mortality occurred in 1990-1992. Dead stems displayed spatially-consistent patterns of decreasing growth after 1985. Comparisons with healthy red oak trees revealed a long-term relative growth decline, and relatively lower actual growth after 1969. Radial growth shocks and decreases during 1980-1993 were associated with weather extremes and defoliating insect outbreaks. A persistent, synchronous growth decline began after 1989 in concert with severe gypsy moth defoliation. Greater mortality in certain stands, along with previously discovered disparities in radial growth and tree-ring response to climate, may be related to historic air pollution from Johnstown, a nearby urban/industrial area.

## INTRODUCTION

Episodes of tree decline and mortality are well-known phenomena that have historically occurred within various forest ecosystems (e.g., Millers and others 1989). Less well-understood are the precise causes of these episodes, yet the question of causal factors becomes especially important as issues of forest health and sustainable forestry assume greater significance to forestry professionals and the public.

Often, tree declines are only noted after they are well underway, and perhaps only after considerable mortality has occurred. A clearer understanding of the nature and potential causes of decline/mortality episodes would be possible if researchers had specific data and observations before and during the decline/mortality process. Lacking this, reconstruction of some events and conditions may be possible by combining tree-ring analyses with available information on climate, pest/pathogen and other potential stress events within the region, if not the affected stands (McClenahen 1995). Tree-ring variation can chronicle the decline process in terms of tree growth, a surrogate for tree health, that can be explored within the context of presently healthy, outwardly-declining, and dead trees within a population (Tainter and others 1990).

This study was aimed at elaborating the timing and nature of environmental stress events and conditions associated with an observed decline/mortality episode on Laurel Ridge in west central Pennsylvania. Considerable dieback and mortality among the oaks, especially northern red oak (*Quercus rubra* L.), has been observed since the early 1980's on Laurel Ridge. Although this trend in declining forest health was associated with a variety of environmental stress events, especially climate extremes and insect defoliations, the potential role of emissions from regional power generation and local industry in Johnstown cannot be overlooked. A general goal of our study was to determine whether local air pollutant emissions may be a factor in the oak dieback and mortality. Our approach was to assess

[1]Associate Professor Emeritus, School of Natural Resources, The Ohio State University's Ohio Agricultural Research and Development Center, Wooster, OH 44691.
[2]Professor Emeritus, School of Forest Resources, The Pennsylvania State University, University Park, PA 16802.
[3]Professor, Department of Plant Pathology, The Pennsylvania State University, University Park, PA 1680.

the amount and extent of recent tree mortality, and to develop tree-ring chronologies on Laurel Ridge from standing dead northern red oaks in stands where live red oak chronologies had been previously obtained (ca. 1987). Our objectives were to estimate the amount and timing of recent red oak mortality within these stands, and to assess the possible causes of mortality by evaluating and comparing temporal and spatial growth responses of trees that have died, live (healthy) trees, and documented stress events in the region.

## METHODS

### Study Area

The study was conducted on Laurel Ridge, an anticlinal ridge of 700-800 m elevation within the Allegheny Plateau Physiographic Province in western Pennsylvania (Figure 1). The climate is continental with a mean growing season temperature of 19.8° C and annual precipitation averaging 104 cm. Soils are well to moderately well drained, very stony loams and silt loams, predominantly Typic Dystrochrepts, occasionally including some Typic Hapludults and Aquic Fragiudults. Forests on Laurel Ridge are second growth mixed hardwoods dominated by northern red oak and other oak species (*Quercus* spp.) on the ridgetop and southerly slopes. Coves and northerly slopes additionally contain black cherry (*Prunus serotina* Ehrh.), red maple (*Acer rubrum* L.), and yellow-poplar (*Liriodendron tulipifera* L.). Fire apparently was an important influence on vegetation of Laurel Ridge at least from colonial times (Harpster 1938) until effective suppression efforts in recent decades. Other factors that strongly influenced present forest composition and condition include the loss of American chestnut (*Castanea dentata* (Marsh.) Borkh.). By chestnut blight, extensive logging for timber and charcoal wood, periodic weather extremes, insect defoliations, and over-browsing by deer.

Historically, $SO_2$ has been a major air pollutant within the vicinity of Conemaugh Gap (Figure 1) since the late 19th century when iron and coke production began in the greater Johnstown area (Brown 1989). Power generating stations on the west side of Conemaugh Gap began operation in 1950 (Seward station, 201 MW generating capacity) and in 1970-71 (Conemaugh station, 1700 MW generating capacity) (Hutnik and others 1989).

### Field Procedures

Healthy red oaks were originally cored within 10 stands (1-10 ha) on Laurel Ridge, mostly on upper west slopes and ridgetops, between 1987-1992 (Figure 1). A subsequent tree mortality survey and core sampling of standing dead red oak stems within the same stands in 1994 is the focus of this paper. To estimate the amount of recent tree mortality, 5-7 variable (10 BAF prism) plots were located at 50-m intervals along transects through each stand in 1994. All standing tally trees in the dominant, codominant and intermediate crown classes, living and dead, were measured. Species, crown class, dbh (diameter at 1.37 m height), and live or dead status were recorded. Signs of dieback were noted for live trees, although these data are not consistent because most stands were sampled in October.

Pairs of 5-mm diameter tree cores were extracted from up to 10 standing dead red oak stems (138 cores from 69 trees) that had occupied dominant or codominant crown positions. Nearly all dead candidate trees were sound enough to permit coring. Dead stems identified by the 10 BAF prism at the plot points were sampled if possible, or elsewhere within the stand as necessary; however, it was not possible to locate 10 dead stems anywhere within some stands. The same data collected for the mortality survey were also recorded for cored trees.

Healthy dominant and codominant red oaks (20-22 trees per stand) had been similarly cored and the ring widths were crossdated and measured ca. 1987 as part of an earlier study of growth relationships with climate and air pollution (manuscript prepared). Those tree-ring data are used here to compare the growth of then-healthy red oaks with the growth history of trees that subsequently died. Trees cored in 1987 were not labeled, so it was not possible to determine for certain which, if any, were part of the dead cohort sampled in 1994. Since increment borer wounds were not found on any of the sampled dead stems in 1994 (some were noted on living trees), it appears that few, if any, of the originally cored (healthy) trees had died by 1994.

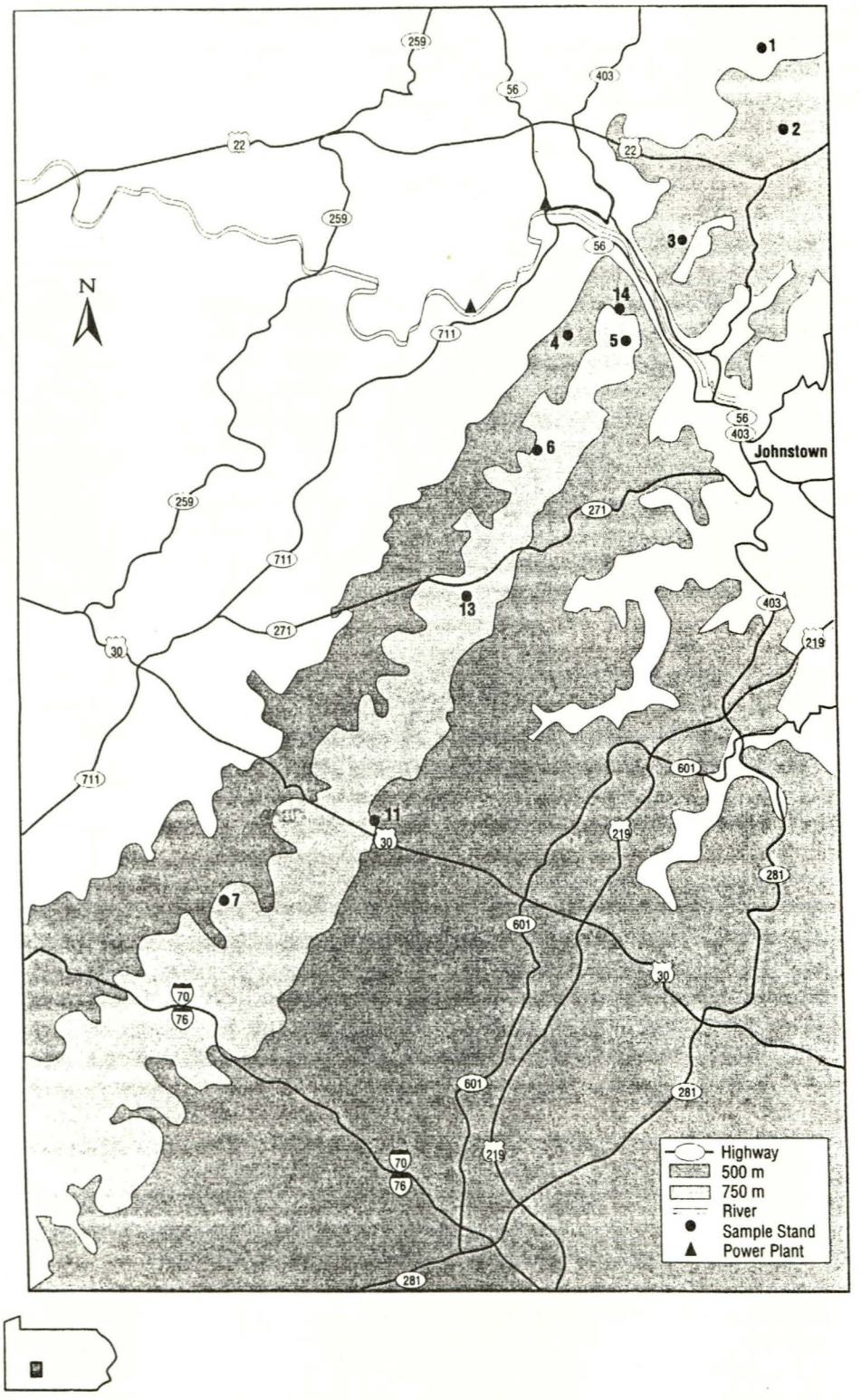


Figure 1. The study area showing numbered sample stands. Scale is 1cm = 3 km.

Annual, systematic observations of forest health, weather-related stress events, $SO_2$ symptomology on bioindicator species, and the status of pests and pathogens have been conducted by the second and third authors (and their predecessors) several times throughout each growing season since 1968 at selected locations on Laurel Ridge and elsewhere in the region. The records of these observations relevant to Laurel Ridge are summarized here from the unpublished annual reports to provide insights about possible causal agents relating to tree growth and mortality patterns. For the surveys of $SO_2$ leaf symptoms, planted (e.g., *Pinus sylvestris* L., *P. strobus* L., *Pseudotsuga menziesii* (Mirb.) Franco) and native (e.g., *Betula lenta* L., *Rubus* sp.) species known to be sensitive to $SO_2$ were examined several times each growing season at the same sites.

### Laboratory Procedures

Densities (stems/ha) of live and dead stems were computed and summarized from the prism plot data for each species. The air-dried tree cores were glued onto labeled wooden holders, surfaced with a series of successively finer sandpapers, and crossdated visually to assure correct assignment of ring years. Ring widths were measured to 0.01 mm precision, and the visual crossdating was verified using a computer program (COFECHA, Holmes 1983).

The last growth year was estimated for each dead tree on the basis of the two cores. This year was taken as the best estimate of the year preceding death. Ring width measurements made on each crossdated core from both live trees cored ca. 1987 and dead trees cored in 1994 were analyzed to evaluate the growth patterns prior to death and in relation to growth behavior of apparently healthy red oaks. First, mean stand basal area increment patterns from dead stems were computed and plotted to observe general growth patterns and trends in relation to known stress events and the timing of mortality. Second, mean annual basal area increment differences between the live and dead red oaks were compared graphically to identify widespread relative discrepancies in growth behavior. The resulting time series of growth difference was subjected to intervention detection analysis (Box and Jenkins 1976, Downing and McLaughlin 1990, McClenahen 1995) to identify significant ($P = 0.20$) relative step or pulse types of growth shocks (i.e, negative growth interventions) experienced by trees that later died.

## RESULTS

### Mortality Survey

Total live stem density varied among stands, from stand 7 with 181 stems/ha to stand 6 with approximately 97 stems/ha (Table 1). However, the lower densities are partly a result of recent mortality (e.g., stands 6 and 2). Northern red oak constituted a high proportion of the canopy stems in most of the sampled stands, but the numbers of dead northern red oak stems ranged from none (stands 4, 13 and 14) to >32 stems/ha (Table 1). On average, red oaks comprised >68% of the dead stems.

Where oak species other than northern red oak were present, they also experienced high percentages of mortality (Table 2). Non-oak species comprised one-half or less of the composition of the sampled stands, and percent mortality was generally less than that of the oaks. Stands 7 and 13 had comparatively large components of black cherry and, in stand 13, less northern red oak. These stands are on relatively broad, concave ridgetops that are slightly more mesic than the others. The highest percentages of dead northern red oaks tended to occur in ridgetop stands near to, or downwind (northeast) of, the Conemaugh Gap area where the power generating stations and Johnstown are located (Figure 2). However, no northern red oak mortality was recorded in stands 4 and 14 which lie on mid- to lower slope positions nearest to the generating stations, or in stand 13 located slightly upwind and on a relatively mesic ridgetop site with comparatively few northern red oaks. Dominant and codominant stems of other species showed a geographical pattern of mortality similar to that of northern red oak, with no mortality again recorded in stands 4, 13, or 14. As noted above, many of the same stands with high northern red oak mortality also had relatively high percentages of dead trees of other species. This further suggests that trees in these northeastern stands may have been exposed to greater stress (with the exception of the more mesic stands).

Table 1. Mean stocking of live and dead dominant and codominant trees and northern red oaks in the sampled stands.

| Stand No. | Canopy Trees | | Northern Red Oak | |
|---|---|---|---|---|
| | Live | Dead | Live | Dead |
| | -no. of stems/ha- | | | |
| 1 | 128.3 | 45.8 | 34.0 | 12.9 |
| 2 | 104.0 | 43.3 | 47.8 | 32.1 |
| 3 | 125.4 | 29.8 | 97.9 | 20.7 |
| 4 | 120.8 | 0.0 | 51.7 | 0.0 |
| 5 | 127.4 | 26.1 | 116.8 | 23.0 |
| 6 | 97.3 | 29.3 | 89.0 | 29.3 |
| 7 | 181.0 | 6.5 | 72.6 | 3.1 |
| 11 | 155.1 | 18.4 | 93.4 | 14.7 |
| 13 | 144.3 | 0.0 | 37.7 | 0.0 |
| 14 | 137.9 | 0.0 | 56.7 | 0.0 |
| **Mean** | **132.2** | **19.9** | **69.8** | **13.6** |

As trees increase in size, they become more susceptible to stress in part due to an increasing ratio of respiring tissue to photosynthetic tissue (Mueller-Dombois, 1992). A t-test of mean dbh between dead and live canopy northern red oaks over all stands (n = 233 and 60, respectively) revealed that the dead trees were significantly (P = 0.003) larger by an average of 6.0 cm (mean dbh = 51.5 and 45.4 cm). This difference was most evident in stand 6, where all of a cohort of larger, older northern red oaks had died. These larger trees were apparently residuals from the previous stand. Ring counts ranged between 102 and 184 years, but heart rot generally precluded age determinations. This supports the general belief that older and larger trees disproportionately succumb to stresses that induce widespread mortality (Mueller-Dombois 1992).

Tree-Ring Data

A total of 132 cores from 69 standing dead red oaks in nine stands were successfully crossdated and the ring widths measured. At least one core from each sample tree was crossdated (95.6% of all cores). No dead northern red oaks were found for sampling in a tenth stand (stand 13), and only one specimen was obtainable in stand 4. The outermost ring was nearly always present on at least one of the two cores extracted from each tree (extensively decayed stems were not sampled, but such stems were rare). Absent rings are another potential problem in establishing the year of death (Mast and Veblen 1994). We did not independently test for the occurrence of missing rings among northern red oaks approaching death. However, unlike some coniferous species (Mast and Veblen 1994, McClenahen and McCarthy 1990), it is unlikely that northern red oaks are capable of surviving more than a year or two in so moribund a state. Although the total chronology period for most stands extended back into the nineteenth century, the selected common analysis period beginning in 1920 often included only about half of the cores, or less. In fact, the distribution of approximate dead tree ages (earliest ring) shows that most of the dead trees were around 70-80 years old (median age 77 years), but with 19% over 100 years old (Figure 3). These age distributions generally reflect the ages of trees in the sampled stands, along with the tendency for older and larger trees to disproportionately die as noted earlier.

Table 2. Average numbers of dominant and codominant stems and percent dead stems by species in sampled stands. Stands are sequenced from north to south on Laurel Ridge.

| | Stand Number | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 14 | 5 | 4 | 6 | 13 | 11 | 7 |
| Northern RedOak | | | | | | | | | | |
| Stems/ha | 46.9 | 79.9 | 118.6 | 56.7 | 139.8 | 51.7 | 118.3 | 37.7 | 108.1 | 75.7 |
| % dead | 27.5 | 40.2 | 17.4 | 0.0 | 16.4 | 0.0 | 24.8 | 0.0 | 13.6 | 4.1 |
| White Oak | | | | | | | | | | |
| Stems/ha | 43.3 | 17.7 | - | - | - | 0.9 | 1.8 | - | - | - |
| % dead | 67.2 | 39.0 | - | - | - | 0.0 | 0.0 | - | - | - |
| Chestnut Oak | | | | | | | | | | |
| Stems/ha | - | 8.6 | 6.4 | 47.8 | - | 27.5 | - | - | 15.2 | - |
| % dead | - | 50.0 | 100.0 | 0.0 | - | 0.0 | - | - | 15.1 | - |
| Black Cherry | | | | | | | | | | |
| Stems/ha | 34.6 | - | 17.6 | 6.3 | 6.7 | - | 5.5 | 94.4 | 9.7 | 90.6 |
| % dead | 0.0 | - | 15.3 | 0.0 | 46.3 | - | 0.0 | 0.0 | 14.4 | 3.8 |
| Red Maple | | | | | | | | | | |
| Stems/ha | 28.2 | 33.1 | 6.8 | 19.2 | 7.0 | 28.7 | - | 12.2 | 38.9 | 19.2 |
| % dead | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | - | 0.0 | 0.0 | 0.0 |
| Other Hdwds | | | | | | | | | | |
| Stems/ha | 21.1 | 8.0 | 5.8 | 7.9 | - | 11.9 | 1.0 | - | 1.6 | 2.0 |
| % dead | 18.0 | 0.0 | 0.0 | 0.0 | - | 0.0 | 0.0 | - | 0.0 | 0.0 |
| Totals | | | | | | | | | | |
| Stems/ha | 174.1 | 147.3 | 155.2 | 137.9 | 153.5 | 120.7 | 126.6 | 144.3 | 173.5 | 187.5 |
| % dead | 26.3 | 29.4 | 19.2 | 0.0 | 17.0 | 0.0 | 23.1 | 0.0 | 10.6 | 3.5 |

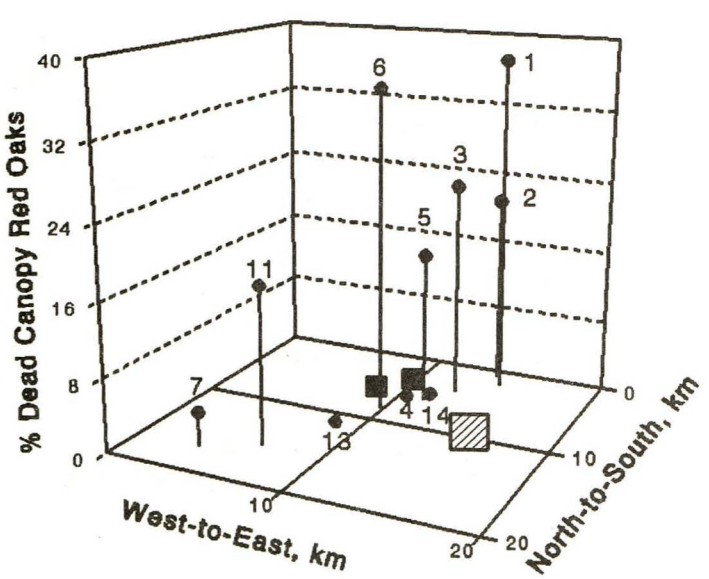


Figure 2. Percentages of dead dominant and codominant red oaks in sampled stands. Solid squares indicate power generating stations, the crosshatched square indicates the location of Johnstown.

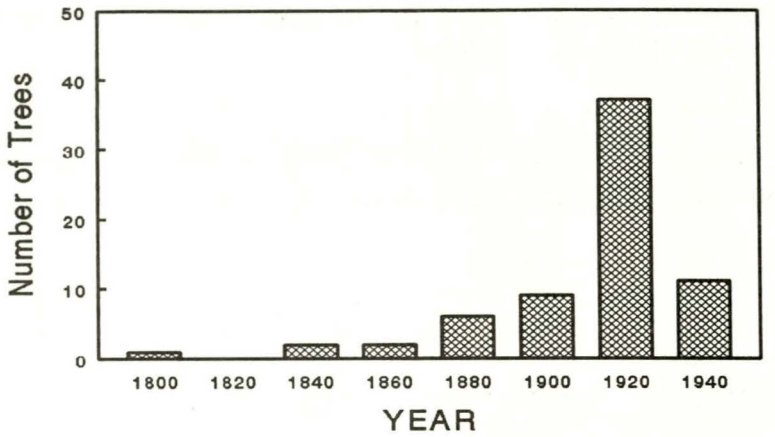


Figure 3. Distribution of approximate pith years for cored dead northern red oaks.

Figure 4 reveals that the distribution of last growth years among all sampled trees peaked in 1990 and 1991. It is difficult to know in exactly which years these trees would have appeared to be dead relative to the final growth ring since foliage may have died within the last growth year, or the tree may have failed to flush the following year. Thus, observable peak northern red oak mortality on Laurel Ridge should have been evident between 1990 and 1992. Our annual observations of forest health and potential stressors indicate that significant oak mortality and branch dieback first appeared in 1983, a drought year that was preceded by a cankerworm defoliation in 1982 (Table 3). Oak dieback and mortality continued through 1989 and 1990 when gypsy moth defoliations occurred. Our 1991 report stated "significant oak mortality on the ridgetops," which was reported in 1992 as "still ongoing." In 1993, crown dieback of oaks was listed as the most severe disease problem among broad-leaved species, and dieback (but not new mortality) was noted to continue into 1995. Thus, our general observations of oak mortality on Laurel Ridge confirm the tree-ring estimate of peak northern red oak mortality in 1991 and 1992.

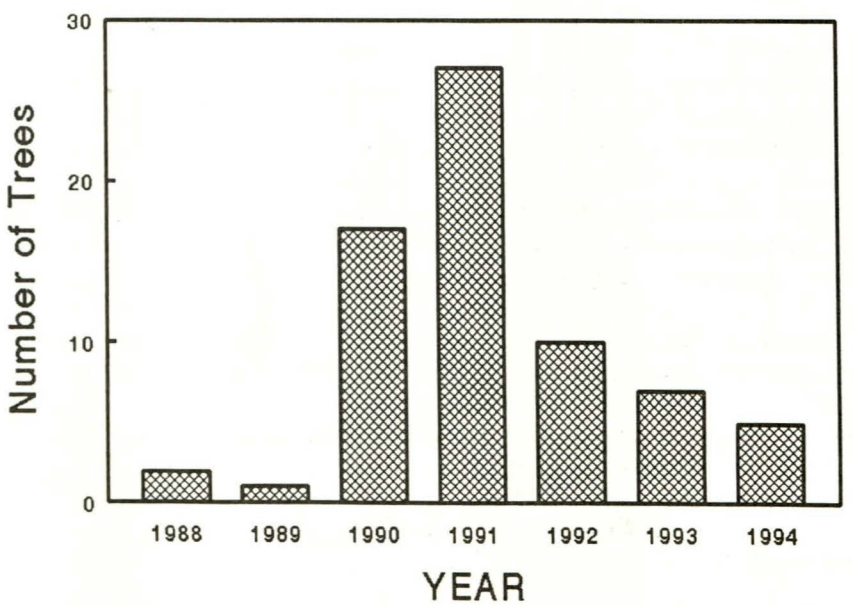


Figure 4. Distribution of last growth years for cored dead northern red oaks on Laurel Ridge.

Table 3. Summary of major forest health and pest/pathogen problems observed within the Laurel Ridge study area from 1968 through 1994 (years with no unusual problems omitted).

| YEAR | EVENTS |
|---|---|
| 1968 | Periodic (17-year) cicada outbreak. |
| 1969 | Localized forest tent caterpillar defoliation. |
| 1970 | Localized forest tent caterpillar defoliation centered in the vicinity of stand 3 (100% defoliation); defoliation also probable in stands 4, 5, 6, and 14. Conemaugh Unit 1 on line in May; $SO_2$ injury recorded on bioindicator species (*Betula lenta* L. and others) near stand 3. |
| 1971 | Severe forest and eastern tent caterpillar defoliation centered near stand 3, defoliation probable in stands 4, 5, 6, and 14. Conemaugh Unit 2 on line in May; $SO_2$ injury on bioindicator species near stand 3. |
| 1972 | Moderate to severe leaf tatter on maple and oak due to Hurricane Agnes, especially at stand 6 and other exposed ridgetop sites. Severe regional frost the first week of June. |
| 1973 | Relatively lower $SO_2$ symptom severity on bioindicator species near stand 3. |
| 1974 | Relatively severe $SO_2$ symptoms on bioindicator species near stand 3. |
| 1975 | No unusual conditions reported; the "...entire study area [was] relatively free of defoliating insects." Relatively moderate $SO_2$ symptom severity on bioindicator species near stand 3. |
| 1976 | Insect injury minimal. Severe frost in mid-May after exceptionally warm April and early May; no mention of frost injury specific to oak. Frost injury worst at higher elevation. No $SO_2$ symptoms on bioindicator species near stand 3; extensive $SO_2$ injury on native species downwind from generating stations (an $SO_2$ episode occurred in June). |
| 1977 | Weather extremes: alternating cold and warm periods in February, warm and dry in April, cool and wet in early May, (>32° C in mid-May followed by a late May frost). |
| 1980 | Moisture stress in late summer. |
| 1982 | Severe cankerworm defoliation, including oaks on Laurel Ridge. Unusual weather: cold, snowy winter, late spring, warm and dry mid-April through mid-May, cool and rainy into June, hot summer with below average precipitation. |
| 1983 | Considerable top dieback and tree mortality on Laurel Ridge, primarily on ridgetop. Severe anthracnose on oak leaves. A drought year. |
| 1984 | Shot-hole feeding injury severe on leaves of oak and black cherry on Laurel Ridge. Trace $SO_2$ injury on sensitive species near stands 3, 4, 14, and 6. |
| 1985 | Periodic (17-year) cicada the most severe foliar injury; shot-hole feeding injury also prevalent on northern red oaks and black cherries. |

*(Table 3 continued)*

*(Table 3 continued)*

| YEAR | EVENTS |
|---|---|
| 1986 | An early spring and light June frost caused no apparent damage. Shot-hole injury common on oaks and black cherry. Dead branches from 1985 cicada injury evident. Oak dieback and mortality on Laurel Ridge continued. |
| 1987 | Mid- and late-summer heat waves, followed by cooling that prematurely ended the growing season in early autumn. Shot-hole injury common on oaks and black cherries. |
| 1988 | Oak dieback and mortality continues on Laurel Ridge. Severe drought the last 3 weeks of June and first 2 weeks of July. Drought most severe in the north and decreasing in the southern part of the study area. High temperatures from June through August, interspersed by low temperature events, one in June with temperatures of -1 C. |
| 1989 | Severe defoliation by gypsy moth, ca. 100% in some areas of Laurel Ridge. Oaks largely refoliated by autumn. |
| 1990 | Severe gypsy moth defoliation, especially on Laurel Ridge; high incidence of top-dieback noted on oaks. Dormant-season temperatures fluctuated greatly. |
| 1991 | Severe oak dieback and mortality on Laurel Ridge. More mortality, dieback, and sparse foliage among oak (and other species) than in 1989-1990. Light gypsy moth defoliation. Extremely hot from May through autumn, with 50% below-normal precipitation. |
| 1992 | Continued high oak mortality and dieback, especially on Laurel Ridge. Light to no gypsy moth defoliation. Cool growing season, good growing conditions. |
| 1993 | Continued severe, widespread oak dieback and mortality. Insect defoliation light. Comparatively minimal $SO_2$ injury. Hot, somewhat dry growing season |

The tree ring data from dead stems are probably only indicative of trees that died in the previous 6-7 years; stems from earlier mortality may be too decayed for core sampling. However, very few stems in the sampled stands were not cored due to decay, suggesting that the mortality patterns in Figure 4 are probably representative of the years since about 1988. Regardless, our cores from dead trees would not likely have included the initial oak mortality that we observed ca. 1983.

The mean basal area growth patterns of dead northern red oaks are shown for each stand in Figure 5. A severe mid-1960's drought that occurred throughout the Northeast represents one of the most pronounced and widespread growth shocks that occurred on Laurel Ridge. This growth shock was also generally prominent in the healthy trees cored ca. 1987. However, growth of dead northern red oaks in several stands was either not severely impacted by this drought, or growth recovery occurred immediately afterward (viz., stands 1, 2, 3, 4, and 14). Conversely, growth of trees in stands 5, 6, 7, and 11 failed to fully recover and a general decreasing growth trend followed. It is interesting to note that stand 5, which may have been most exposed to Johnstown pollutant emissions, subsequently exhibited an increasing growth trend beginning in the late 1970's and early 1980's coincident with dramatic reductions in the Johnstown steel industry (Brown 1989).

Another notable feature of the growth trends for individual stands is the widespread, negative growth shocks apparent in the early 1970's (Figure 5). These were seen mostly in either 1971 (stands 1, 4, and 14) or 1972 (stands 2, 3, 5, 6, 11, and perhaps 7). These growth shocks were also evident among the live northern red oak chronologies. The timing of these growth shocks approximately coincided with several potential stress events: (1) the startup of

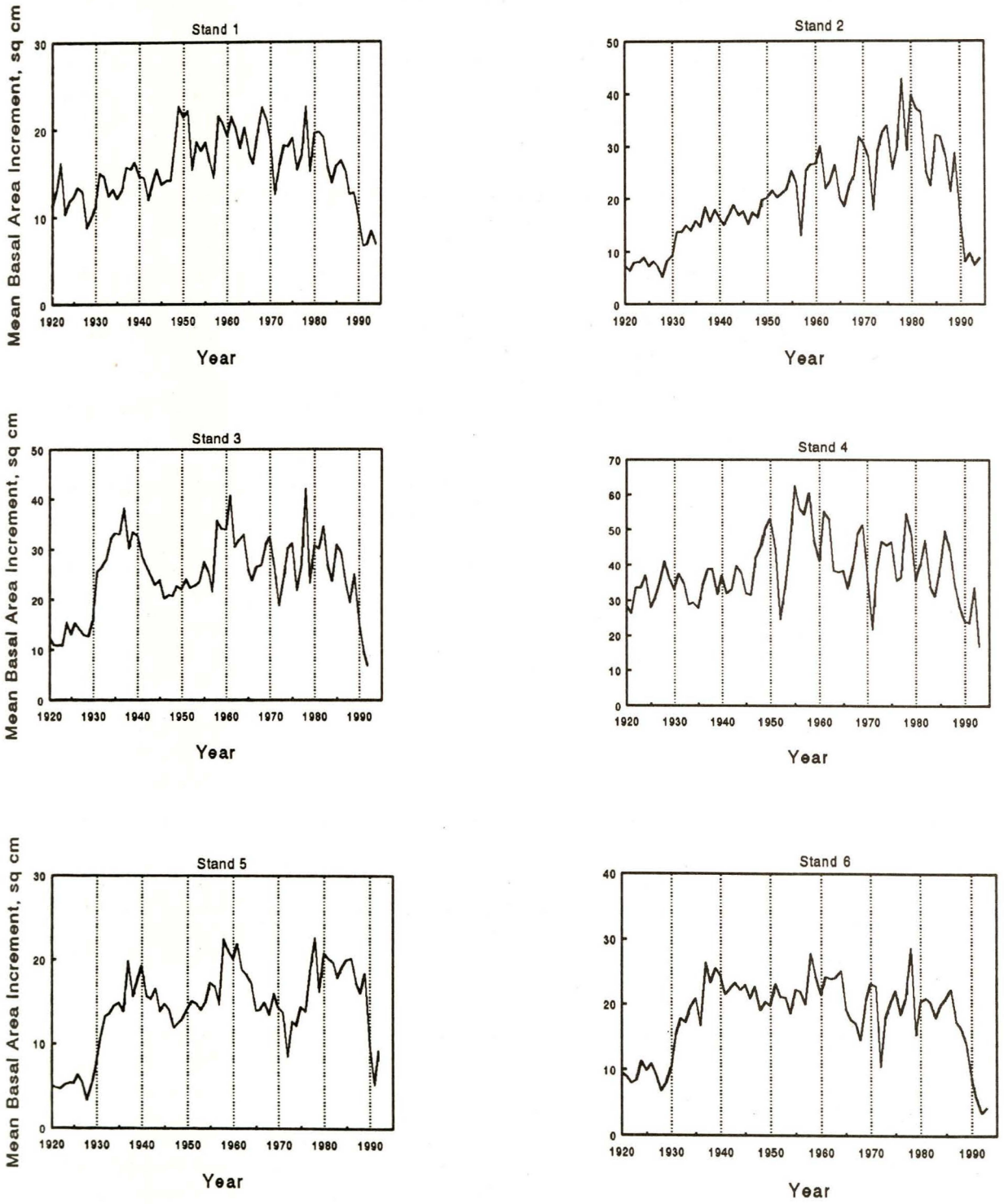


Figure 5 (continued)

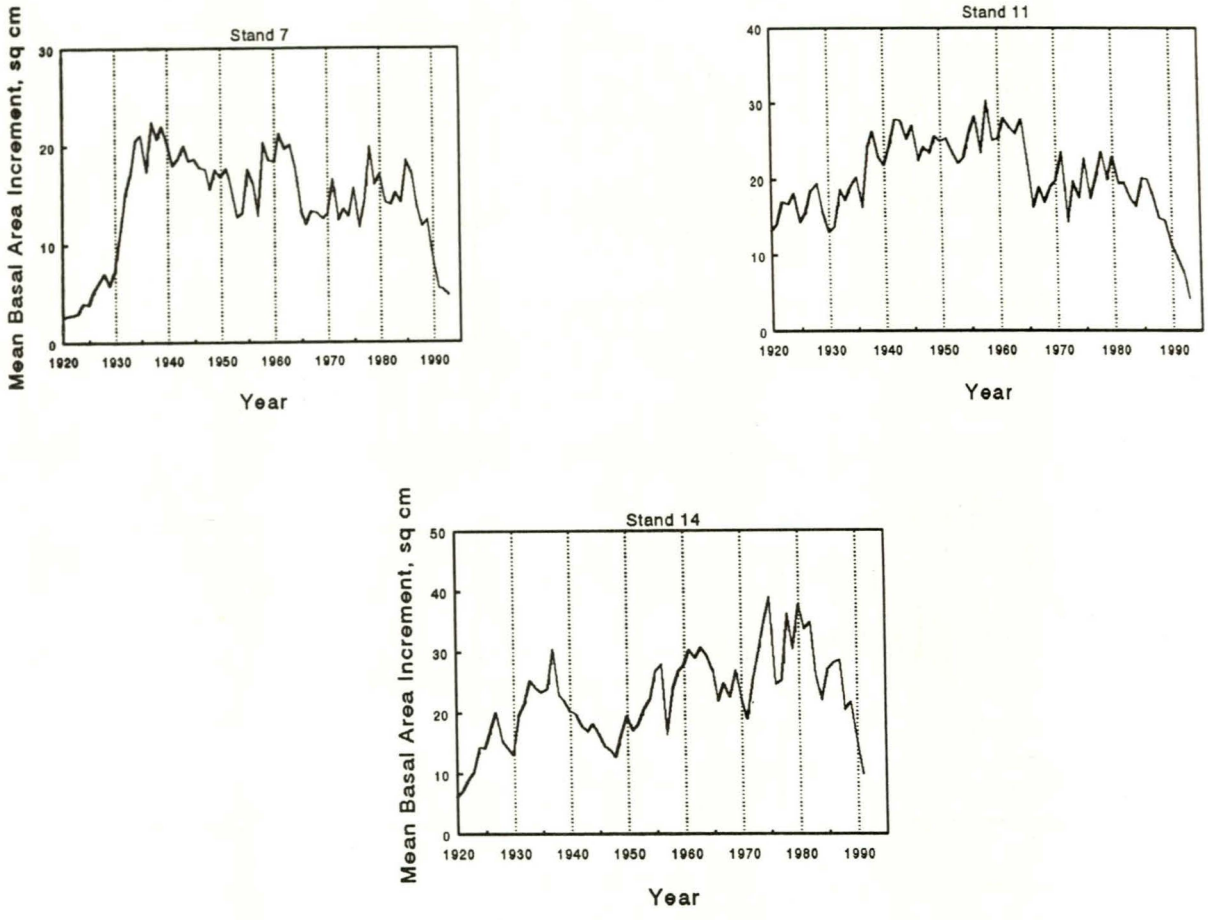


Figure 5. Mean stand basal area growth patterns of cored dead northern red oaks.

the Conemaugh power station in 1970 and 1971, (2) the initial, localized appearance of $SO_2$ symptoms on leaves of some sensitive species on Laurel Ridge, (3) a series of forest and eastern tent caterpiller outbreaks of varying defoliation severity and spatial distribution on Laurel Ridge in 1969 and especially 1970-1971 and, (4) several weather-related stress events (Table 3).

Most of the dead northern red oak chronologies reveal sharply decreasing growth trends beginning in 1988, a year of severe drought, until death occurred (Figure 5). Interestingly, some stand chronologies showed slight, temporary growth recovery in 1989 despite the severe gypsy moth defoliation in that year.

Evaluation of the time series of mean growth trend differences between relatively healthy (cored ca. 1987) and dead northern red oaks (cored in 1994) was conducted to establish the timing of major and widespread growth shocks (i.e., step and pulse interventions) that registered more severely among the dead northern red oaks on Laurel Ridge (Figure 6). Early growth of the dead northern red oaks, relative to the healthy trees, decreased consistently from 1920 until around the 1950's, possibly a result of stand dynamics, an artifact of early sample depth, or a predisposing condition to decline. Intervention analysis shows that growth of the dead northern red oaks was comparatively more strongly reduced by stresses around 1952, 1980, and 1985.

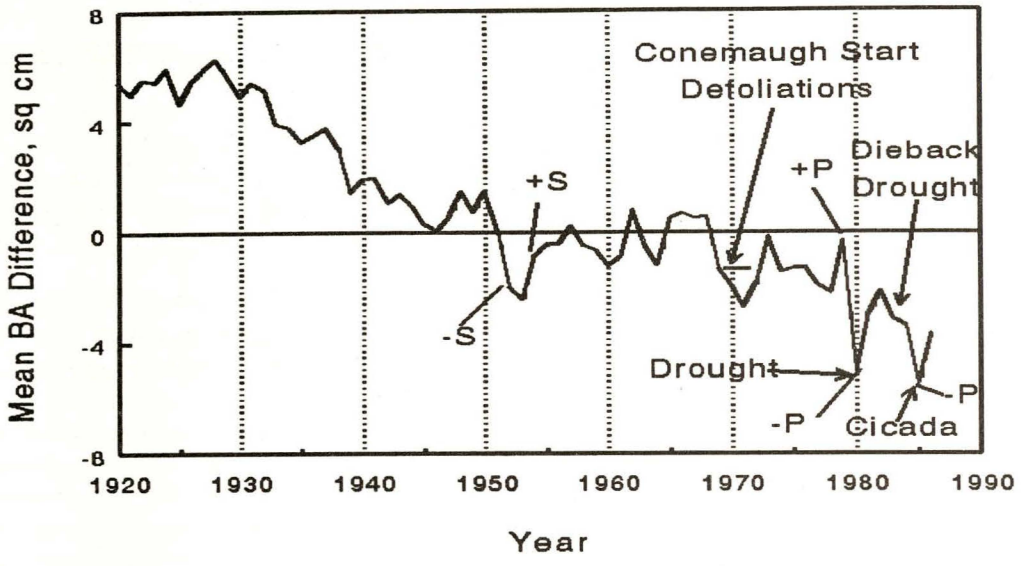


Figure 6. Time series of mean basal area growth differences between dead and live northern red oaks (dead minus live) cored on Laurel Ridge. Significant (P = .20) positive (+) and negative (-) step (S) and pulse (P) growth interventions are indicated.

The temporary step growth decrease in 1952-1953 (Figure 6) roughly coincides with a 1953-1955 drought (mean growing season precipitation deficit of 100 mm), and a series of defoliator outbreaks in the mid-1950's in many areas of Pennsylvania (Nichols 1968). An apparent relative growth shock in 1970-1971 probably due to the tent caterpillar infestation was not significant, suggesting that the growth impact of this defoliation was not very different between the healthy and dead northern red oaks. Conversely, drought in 1980 and a 17-year cicada outbreak in 1985 seem to have affected the growth of the dead northern red oak group more severely than the healthy trees. From these relative patterns of growth behavior, it appears that the dead northern red oaks had been predisposed (after Mueller-Dombois 1992) to growth decline relative to the general population by about 1980. The predisposing factor(s) may have been all or some combination of drought and defoliators in the 1950's, the statistically non-significant but notable growth shocks in 1970-1972, or other general stresses that more or less adversely affected northern red oak growth on Laurel Ridge (such as drought in the 1960's). However, the steady, long-term relative growth decline of the dead cohort from around 1930 until the late 1940's (Figure 6) may be indicative of even earlier predisposing factor(s) such as extreme drought (1931) and/or microsite conditions.

## DISCUSSION AND SUMMARY

Tree-ring analyses, annual field observations of tree health, and a tree mortality survey revealed that a contemporary tree mortality episode on Laurel Ridge was largely confined to oaks, especially northern red oak, the predominant oak species in these ridgetop stands. Between 0 and 40% of northern red oaks were dead in the 10 sampled stands. The variation in percentage of dead trees appeared to be partly related to moisture relationships, with less mortality on relatively mesic and (or) lower slope sites. The episode of high northern red oak mortality began about 1990, peaked in 1991-1992, and decreased through 1994. The amount of 1994 mortality remained greater than pre-episode levels (1988 and 1989), indicating that the episode, though subsiding, had not yet ended.

The chronology of stress events, documented since 1968, allows some inferences regarding factors that may have caused this decline and mortality episode. The 1980 relative growth shock, induced by drought, may have been the inciting factor that initiated declining growth and health among the cohort of trees that ultimately died (Figure 6). Subsequent stresses, such as the defoliation by cankerworm in 1982, drought in 1983 and 1988, periodic cicada injury in 1985 and, finally, gypsy moth defoliations in 1989 and 1990, probably contributed to the spiral of decline

leading to death. We first observed external indication of declining health, in the form of widespread branch dieback, in 1983. The dead tree chronologies in all stands showed a period of precipitous basal area growth decrease beginning after 1985 (Figure 5), implying that the major stress factor(s) were prevalent throughout the Laurel Ridge study area. It would be interesting to know whether the growth and health of declining trees would have recovered in the absence of the 1988 drought and the severe gypsy moth defoliations in 1989 and 1990. Regardless, the succession of stress events beginning around 1980 evidently triggered the sharp growth decrease and widespread crown dieback among northern red oaks on Laurel Ridge that resulted in the high mortality detected by our qualitative and quantitative field observations and from the tree-ring chronologies.

In addition to site factors, geographical patterns suggest that greater northern red oak mortality tended to occur northeast of the Conemaugh Gap, generally downwind of local power generating stations and the industrial city of Johnstown. High mortality in this direction was largely confined to upper elevation stands directly above Johnstown and to the northeast, while stands on the ridge slopes nearest to, and downwind of, the generating stations (on the opposite side of the ridge from Johnstown) had little to no oak mortality. The $SO_2$ concentration patterns in the region are not known. $SO_2$ injury was found on bioindicator species near stand 3 in the early years of the Conemaugh station startup, but symptoms there have been absent since the mid-1970's. We have consistently observed $SO_2$ symptoms on *Rubus* sp. on the ridge slope above Johnstown. We previously reported that leaf sulfur concentrations are greater nearest the generating stations (and Johnstown) in the northeast (downwind) direction (Hutnik and others 1989), while foliar $O_3$ injury on a bioindicator species (hybrid poplar) was unrelated to, or slightly lower near, the generating stations (Davis and others 1993). This suggests that both Johnstown industries and the generating stations have contributed to leaf sulfur concentrations and to localized foliar $SO_2$ symptoms. Another of our studies indicates that historical industrial emissions from Johnstown may have adversely affected northern red oak growth and response to climate variation in some of these stands (manuscript in review). The issue of whether the startup of the Conemaugh generating station in 1970-1971, or the concurrent severe tent caterpillar defoliations, played a role in the negative growth perturbations is being further investigated. Preliminary results revealed that ring-width chronologies of yellow-poplar (a non-host species) lacked the negative growth interventions seen in northern red oak (a host species), confirming that defoliation by insects was the most likely cause of northern red oak growth reductions in 1970-1972. The prevalence of the northern red oak growth intervention throughout the study area lends additional weight to this notion. Thus far, our results do not indicate that the power generating stations had a significant impact on tree health or northern red oak mortality on Laurel Ridge.

## ACKNOWLEDGMENTS

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# ASSESSING SPECIES VARIATION WITHIN FOREST COVER TYPES DELINEATED USING COLOR-INFRARED AERIAL PHOTOGRAPHS

Clarence P. Neese, Jr. and Linda S. Gribko[1]

Abstract: Color-infrared aerial photographs are widely used to aid in the classification and inventory of large forested tracts. Areas of similar photographic color and texture are delineated and a limited number of categories (or groups of areas with similar features) are identified. The forest cover type, and sometimes the average timber volume, of each group is then determined by "ground-truthing" randomly selected representative areas. This technique provides a quick, cost-effective method of forest cover typing and forest inventory across large landscapes. However, there is potential for significant variation in species composition within each classified group. This variation is often ignored; therefore, classifications made using this technique may be too general to be of much value in forest management.

The purpose of this study was to determine what degree of variability can be expected in Appalachian hardwood stands classified using the technique described. Color-infrared aerial photographs of the 3,100-ha West Virginia University Forest were scanned, allowing digital rendition of each photograph. Obvious landmarks on each photo were located in the field and the geographic location of each landmark was determined using a Trimble GeoExplorer GPS unit and base station. Arc/Info software was then used to create a computerized photo mosaic of the Forest. Major differences in tree cover, expressed as variations in color and texture, were identified through supervised classification. Contiguous areas of similar features (stands) were delineated and a small number of general groups were identified. The SAF forest cover type of each categorized group was determined through field inventory of randomly selected stands in each group. All of the stands in the three most prevalent categories were then intensively inventoried to ascertain variation in species composition within these categories. The outcome was an estimation of the range of ground conditions that appear on an aerial photograph as a single forest cover type. It is anticipated that the results of this study will aid in the refinement of this commonly used technique of delineating forest cover types across large landscapes.

[1] Graduate Research Assistant and Assistant Professor, respectively, of Forest Management, Division of Forestry, P.O. Box 6125, West Virginia University, Morgantown, WV 26506-6125.

# HAY-SCENTED FERN SPORE PRODUCTION FOLLOWING CLEARCUTTING

Kathy A. Penrod and Larry H. McCormick[1]

Abstract: Hay-scented fern is a common forest understory weed native to the Appalachian region. It interferes with oak and other hardwood seedling growth and often leads to regeneration failures. Harvesting is know to increase rates of vegetative expansion, spore germination, and possibly spore production of hay-scented fern. To examine the latter effect, a progressive series of five annual clearcuts was made in each year from 1990 to 1994 in a mixed oak stand in central Pennsylvania. Four populations of hay-scented fern in each clearcut and in an uncut control stand were sampled in August 1994. The number of fronds per square meter increased four-fold in the first two growing seasons following clearcutting then decreased to about twice the density of the uncut control stand following five growing seasons. Percentage of fertile fronds declined the first growing season as new fronds emerged then increased continuously each growing season as new fronds matured. Spore production per frond increased four-fold and estimated population spore production increased twelve-fold by the fifth growing season following clearcutting. Spore production increases of this magnitude can lead to large increases in the soil sporebank of hay-scented fern.

[1] Graduate Student and Professor, respectively, School of Forest Resources, The Pennsylvania State University, University Park, PA 16802.

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Pallardy, Stephen G.; Cecich, Robert A.; Garrett, H. Eugene; Johnson, Paul S., eds. 1997. **Proceedings, 11th Central Hardwood forest conference**; 1997 March 23-26; Columbia, MO. Gen. Tech. Rep. NC-188. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 401 p.

Four invited papers, 46 volunteer papers, and 11 volunteer poster summaries presented at the 11th Central Hardwood Forest Conference. Presentation topics included harvesting, economics, forest health, silviculture, ecology, genetics, physiology, and regeneration.

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KEY WORDS: Forest management, forest ecology, silviculture, stand dynamics, forest health, tree physiology, forest economics, harvesting.

Our job at the North Central Forest Experiment Station is discovering and creating new knowledge and technology in the field of natural resources and conveying this information to the people who can use it. As a new generation of forests emerges in our region, managers are confronted with two unique challenges: (1) Dealing with the great diversity in composition, quality, and ownership of the forests, and (2) Reconciling the conflicting demands of the people who use them. Helping the forest manager meet these challenges while protecting the environment is what research at North Central is all about.